

# Science Foundation Chapter 1

## The Dynamic Workings of the Baylands

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## INTRODUCTION

This chapter describes many of the important factors which will govern the future evolution of the baylands. It describes conceptual frameworks for how baylands evolve, particularly in the light of climate change. The importance of sediment supply and topography are highlighted. The geologic and historic record is then examined for clues to how marshes have responded to significant change in the past. Looking to the future, the likely drivers of change that will govern bayland evolution over the next century are examined.

The 1999 Goals Report discusses some of the lessons learned in tidal marsh restoration that occurred in the 1980's and 1990's. With more restoration happening in San Francisco Bay now than ever before, our understanding of tidal marsh restoration processes continues to advance. However, just as important is our increased awareness of the interconnectedness of adjacent habitats. Given the complexity of the tidal conditions and freshwater inputs to San Francisco Bay, the baylands are actually a dynamic continuum of habitats, and drawing boundaries between the functions of the open water bay, intertidal mudflats, tidal marshes, and the adjacent uplands is difficult and sometimes arbitrary. Over the next century we expect climate change and other drivers to create a more dynamic landscape with shifts in the location and nature of these habitats.

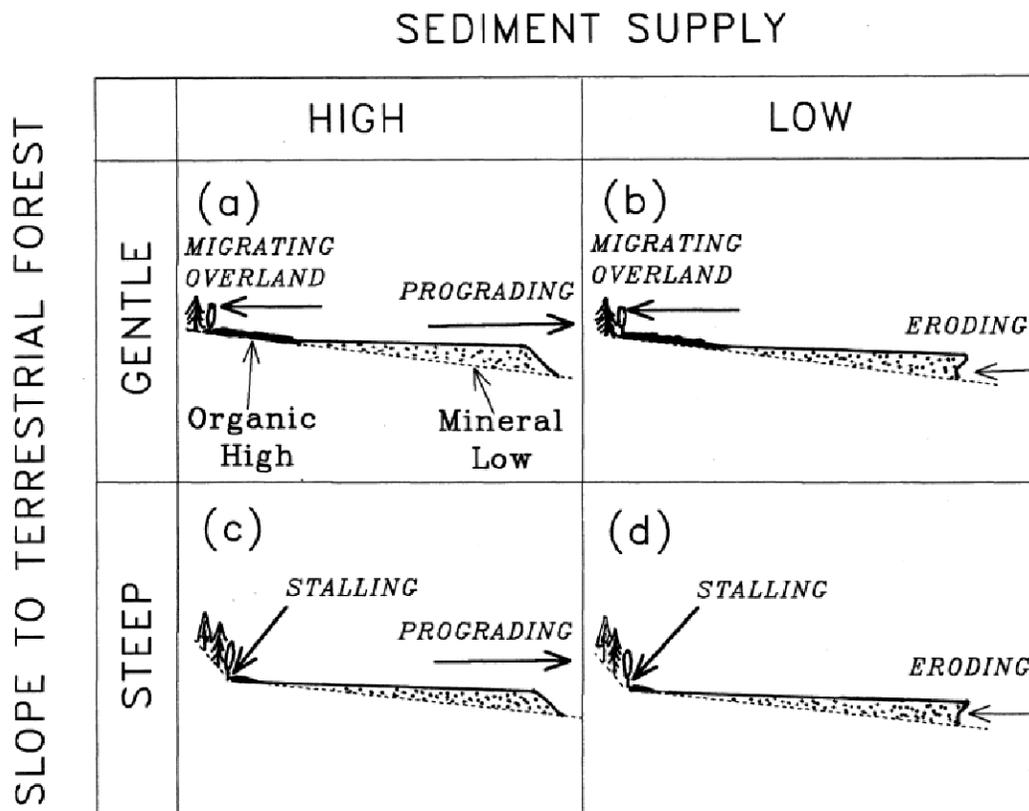
A more accurate way to consider this continuum of habitats involves the concept of a 'complete marsh,' which emphasizes all aspects of a baylands ecosystem and the full gradient of functions and values. Restoring a tidal marsh alone, without considerations of how it connects to the lower elevation mudflats and open water and the higher elevation uplands, may result in an incomplete marsh in terms of ecosystem services. Therefore, this update expands our look at the baylands into the subtidal and upland zones to emphasize the importance of this complete marsh concept and how they may evolve in the future.

## HOW DO BAYLANDS EVOLVE?

Habitat evolution in the light of climate change and sea-level rise in particular, can be related a number of factors which together govern the condition and extent of the habitat:

- migration (also called transgression) - upland migration based on sea level, hydrology, sediment supply, plant processes, topography, and subsidence
- drowning - vertical accretion and change in inundation regime of the marsh surface
- erosion - horizontal/shoreline change along the marsh edge

Migration will in large part be governed by the upland topography landward of the marshes. The concept of ‘elevation capital’, discussed later, provides a framework for describing drowning. Sediment supply and wave energy may be a framework to describe erosional processes. Brinson et al (1995) provide such a framework for addressing the transformation from one habitat class to another as sea-level rises; from uplands through wetlands to mudflat and subtidal. This is illustrated in Figure 1.1 where estuarine-terrestrial interaction can be described as one of four possible interactions based on two conditions at the landward transition zone edge (‘migration’ versus ‘squeeze’) and two at the bayward marsh edge (‘prograding’ versus ‘eroding’).



**Figure 1.1.** Classes of the response of the marshes to sea-level rise as a function of extremes in sediment supply and upland slope, after Brinson et al (1995).

To begin, we will look at factors controlling the evolution of the marsh plain and then look in more detail at shoreline processes and accretion processes.

Because tidal marshes are highly sensitive to elevation, their sustainability will depend on the balance between sea-level rise and marsh sediment accretion (Michener et al. 1997, Morris et al. 2002). As sea-level rises, tidal marshes will have to accrete sediment more rapidly in order to maintain current elevations relative to tidal levels. As with present processes, this vertical accretion of sediment will be a combination of both organic (primarily from plant production) and mineral matter accumulation to offset sea-level rise and vertical land movement (Figure 1.2a).

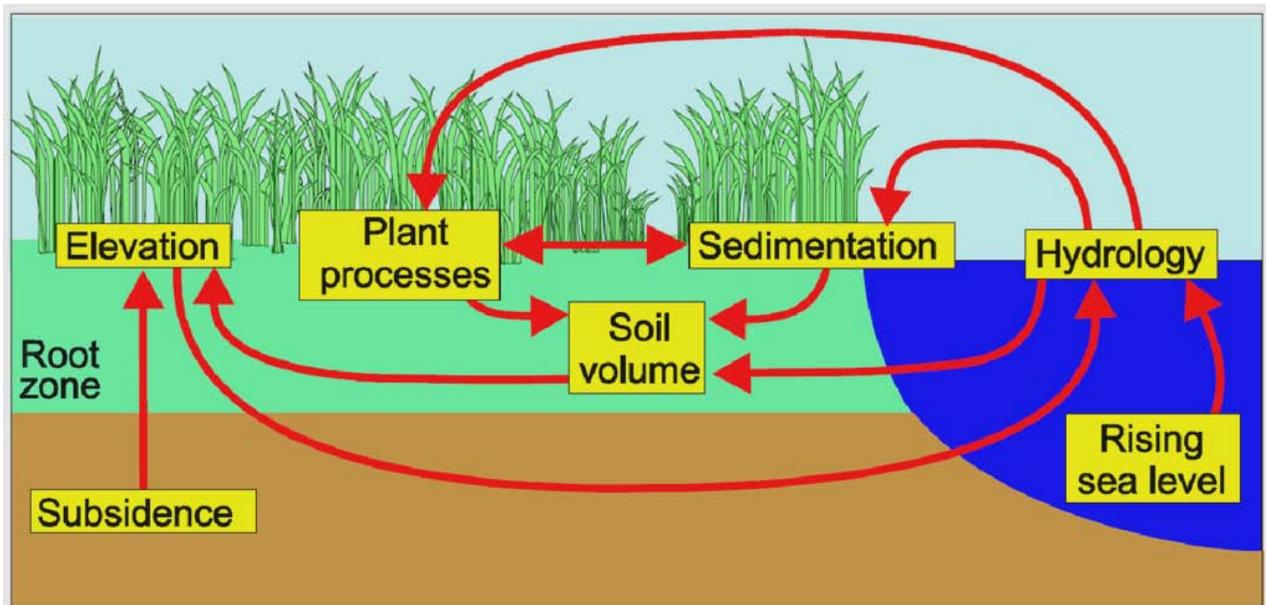


Figure 1.2a. Relationships influencing marsh elevation. Source: Cahoon 1997.

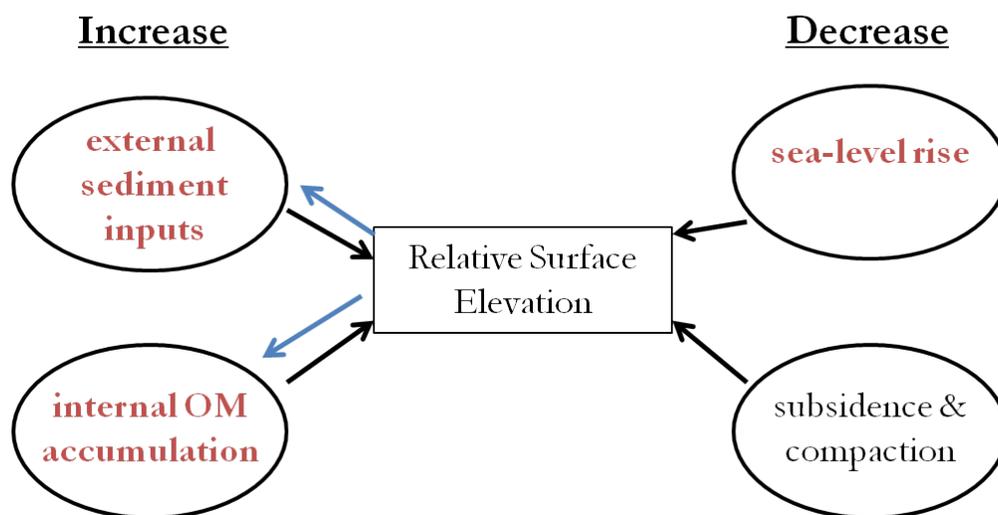


Figure 1.2b. Factors affecting elevation within the marsh. Mineral and organic accumulation as positive factors and sea-level rise, local subsidence, and compaction leading to elevation loss.

There are substantial uncertainties surrounding future predictions of both sea-level rise and suspended sediment concentrations, and with interactions between factors there is even greater uncertainty in predicting sustainability of tidal marshes. If sea-level rise is at the low end of future predictions, and suspended sediment concentrations are not dramatically reduced, it is likely that tidal marsh elevations will remain similar to current conditions (i.e., existing tidal marshes remain highly sustainable, along with relatively rapid development of newly restored tidal marshes). However, if future rates of sea-level rise are closer to predicted high rates, or if suspended sediment concentrations drop substantially, tidal marshes are likely to begin to lose elevation, and restored marshes will develop much more slowly (or not at all, in worst case scenarios). The rate of elevation loss will be influenced directly by the interacting effects of increases in water level and decreases in suspended sediment concentrations (Figure 1.2b).

Most existing Bay tidal marshes are currently dominated by high marsh vegetation and are found at the upper elevation range for tidal marsh ecosystems. Marsh vegetation is directly affected by elevation. It is one of the most important factors affecting frequency, depth and duration of tidal flooding. Site-specific elevations of tidal marsh plants are also affected by exposure and soil type. In general vegetation occurs from just above mean sea level (MSL) to just above mean higher high water (MHHW). *Spartina foliosa* is found at lower elevations, while the marsh plain (at an elevation close to MHHW) is dominated by *Salicornia pacifica*, along with a number of other species depending on local elevation, drainage, soils, site history and other factors (Baye et al. 2000, Grewell et al. 2007). Similarly, frequently flooded brackish marsh sites have characteristic species (e.g., *Schoenoplectus* spp.) with more salt tolerant vegetation on the marsh plain (Vasey et al. 2012). The fact that most Bay tidal marshes are relatively high elevation marsh plains implies that they will remain vegetated even with some loss of elevation. Their relatively high elevation gives them substantial ‘elevation capital’ (Cahoon and Guntenspergen 2010). In other words, they have a large amount of elevation to lose before they are converted to unvegetated mudflats. The concept of elevation capital with specific reference to San Francisco Bay will be discussed in a subsequent section.

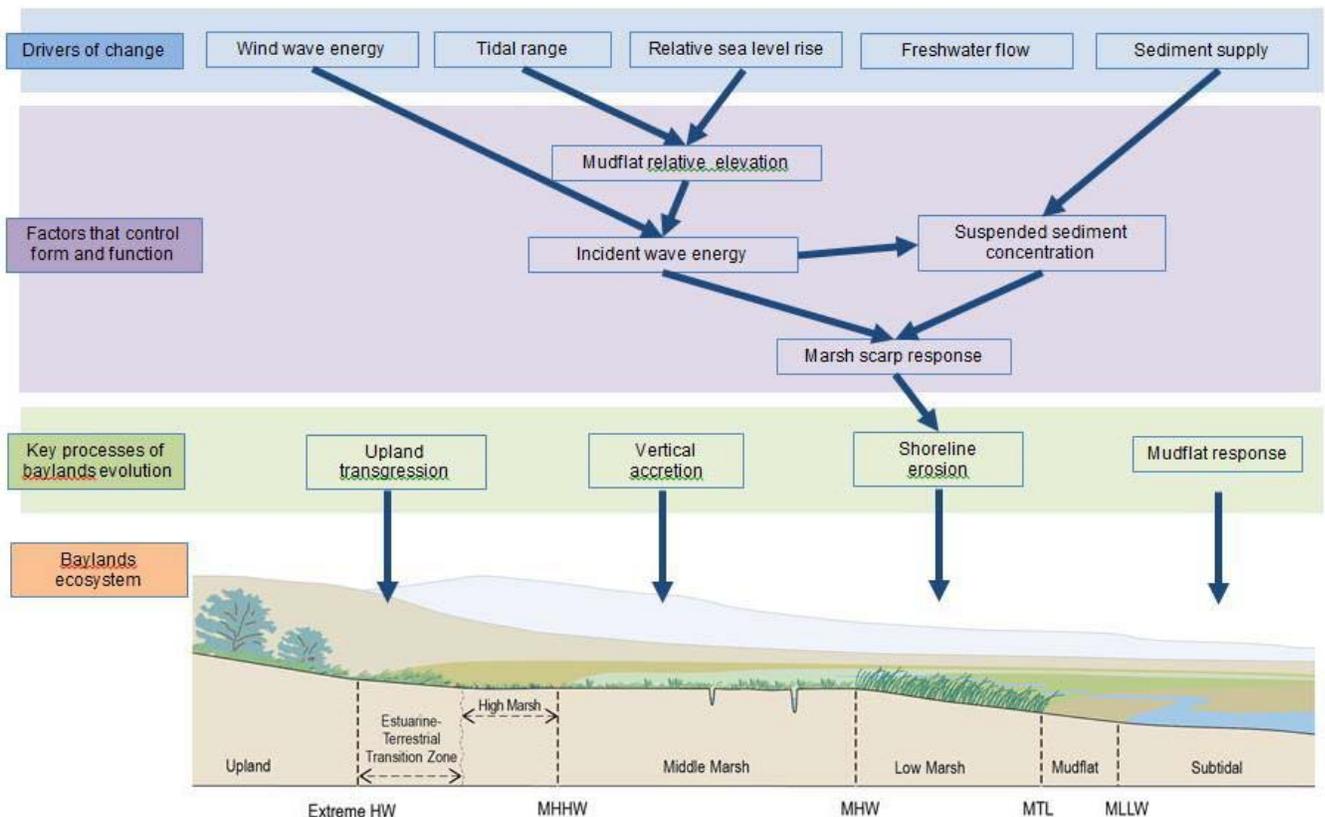
If tidal marshes cannot keep pace with sea-level rise and begin to lose elevation, tidal marsh habitat will be lost and converted to unvegetated mudflats when elevations within the marsh drop below the threshold for survival of marsh vegetation. In places where adjacent areas are relatively flat and at slightly higher elevations, marshes could migrate inland. However, there are areas around the Bay surrounded by levees or other areas with abrupt elevation changes and migration will be constrained. In areas where migration is physically possible, there also are potential limitations depending on land ownership and future land management decisions.

Simultaneous with changes in elevation due to sea-level rise, salinity is likely to increase due to the increased marine influence, as well as to changes in freshwater inputs associated with climate change. Increasing salinity, especially during the summer growing season, will affect vegetation, with a likely shift of more salt tolerant vegetation establishing inland. However, migration of vegetation inland may not be so simple, as there are large differences in soil conditions from freshwater tidal marshes to salt marshes, and dispersal and recruitment of vegetation could be limited. Freshwater wetland soils typically have very high organic matter content and correspondingly low soil bulk density; whereas, salt marsh soils have relatively low organic matter content and much higher soil bulk density (Nyman et al. 1990). Movement of propagules is typically downstream within the estuary (from freshwater marshes to salt marshes), and dispersal and recruitment of vegetation upstream could be much less predictable. As a result of such factors, salt water intrusion can lead to the conversion of freshwater or low salinity wetlands to open water rather than to more salt tolerant vegetation in some systems.

## EROSION

Sea-level rise in the Holocene slowed significantly about 6,000 years ago and the marsh edges of San Francisco Bay have been evolving over the last 2,000 to 3,000 years. While marsh plain evolution has been well documented for San Francisco Bay (Goals Project 1999), less attention has been paid to the low marsh-mudflat transition zone or evolution of marsh edge over time.

The major physical drivers of marsh edge evolution and shoreline change include wind-wave energy and direction, sediment supply, vegetation and sea-level rise (Allen 1989, Schwimmer 2001, Moller and Spencer 2002, Pedersen and Bartholdy 2007) (Figure 1.3). Conditions at the marsh edge are governed by the mudflat which influences the size and energy of waves reaching the marsh – mudflats and marshes are interdependent.



**Figure 1.3.** The major physical drivers of marsh scarp evolution and shoreline change.

Within the normal tidal range, mudflat serves to filter offshore waves to a narrow height band. Small waves increase in height as the water depth shallows and large waves break and are attenuated due to friction (Lacey and Hoover 2011). The result is that waves at the shoreline are relatively constant in height. Mudflat slope and shape may thus control the position of the shoreline. It erodes to widen the mudflat until wave energy is reduced sufficiently so that erosion no longer occurs (this is the same idea as a shore platform/cliff retreat model, e.g., Trenhaile 1987)

A combination of sediment supply and wave energy may control shape and elevation of the mudflat (Bearman et al. 2010), which in turn determines wave energy inshore at the scarp edge. If the mudflat is

high enough, large waves would only reach the shoreline at extreme water levels. If this is the case, scarps may be active for limited periods of time. Where the mudflat is lower, more energy is delivered inshore and wave energy is higher at the shoreline more frequently.

If mudflat does not keep up with sea-level rise, more wave energy will reach the shoreline leading to changes in rate of change and typology. A hypothesis for shoreline evolution of San Francisco Bay is described below, based on a multi-decadal analysis of shoreline change over time in San Pablo Bay and a review of current scientific literature (Figure 1.4).

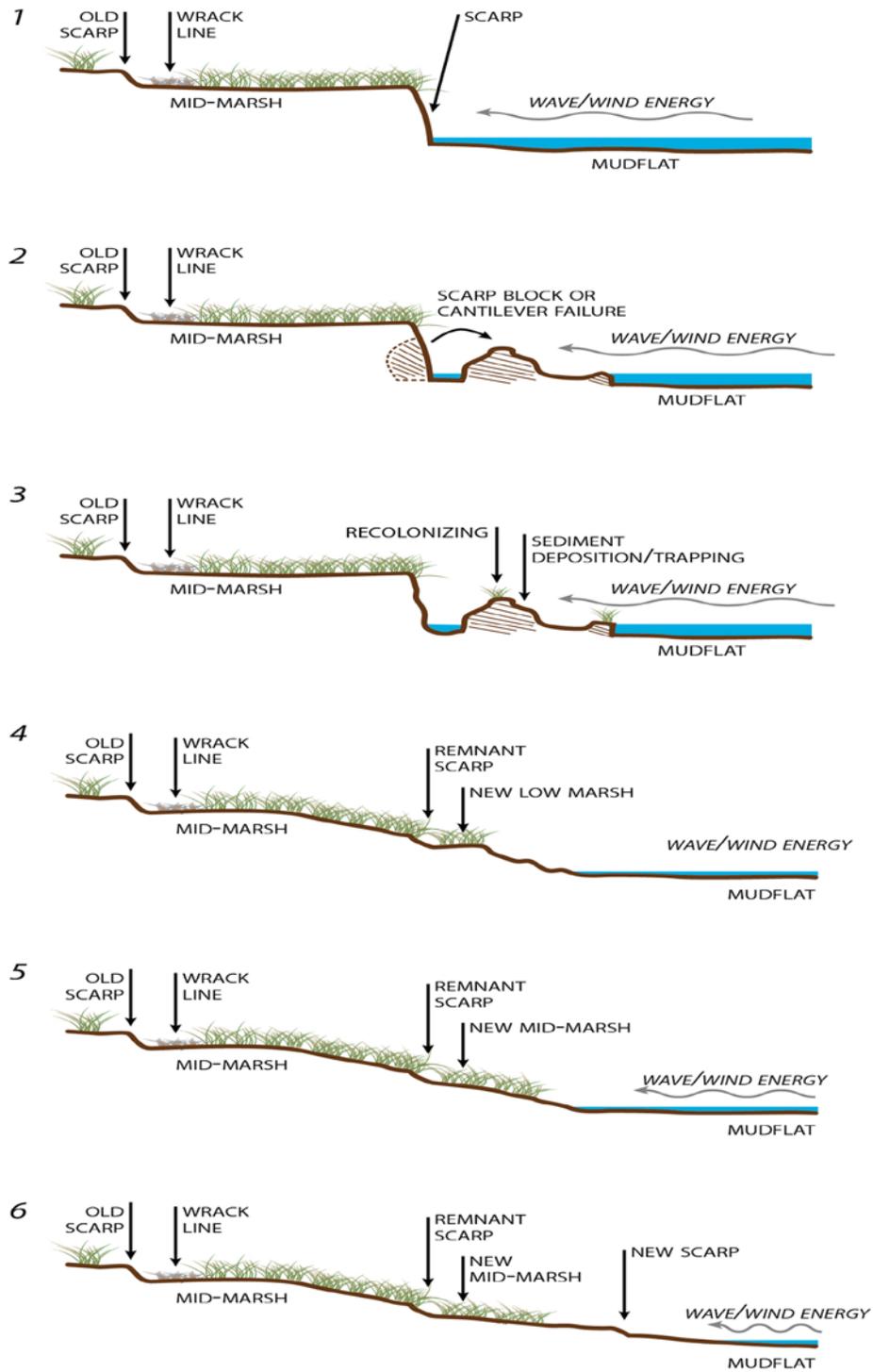
In this conceptual model, a given marsh scarp (1) fails under pressure from wind wave energy and wave run up. Undercut blocks, or cantilever failures, deposit sediment (with or without vegetation) in front of the marsh scarp (2). The block of sediment dissipates wave energy until it is scoured away and redistributed on the mud flat or marsh plan, creating an erosional environment as wave energy is then directed back to the marsh scarp, increasing marsh retreat. If the failure is large enough to redirect wave energy for longer periods of time, the failed blocks may create an environment for sediment deposition and trapping between the old scarp and the failed block (3). A ramped profile begins to fill in the scarp, and build elevation, creating new low marsh, and leaving behind a remnant scarp (4). This ramping continues, and dissipates wave energy such that the low marsh vegetation traps sediment, building up to mid-marsh habitat (5). When the new mid-marsh levels, the ramped profile steepens and wind wave energy begins to erode the new mid-marsh creating a new scarp (6) and the cycle continues.

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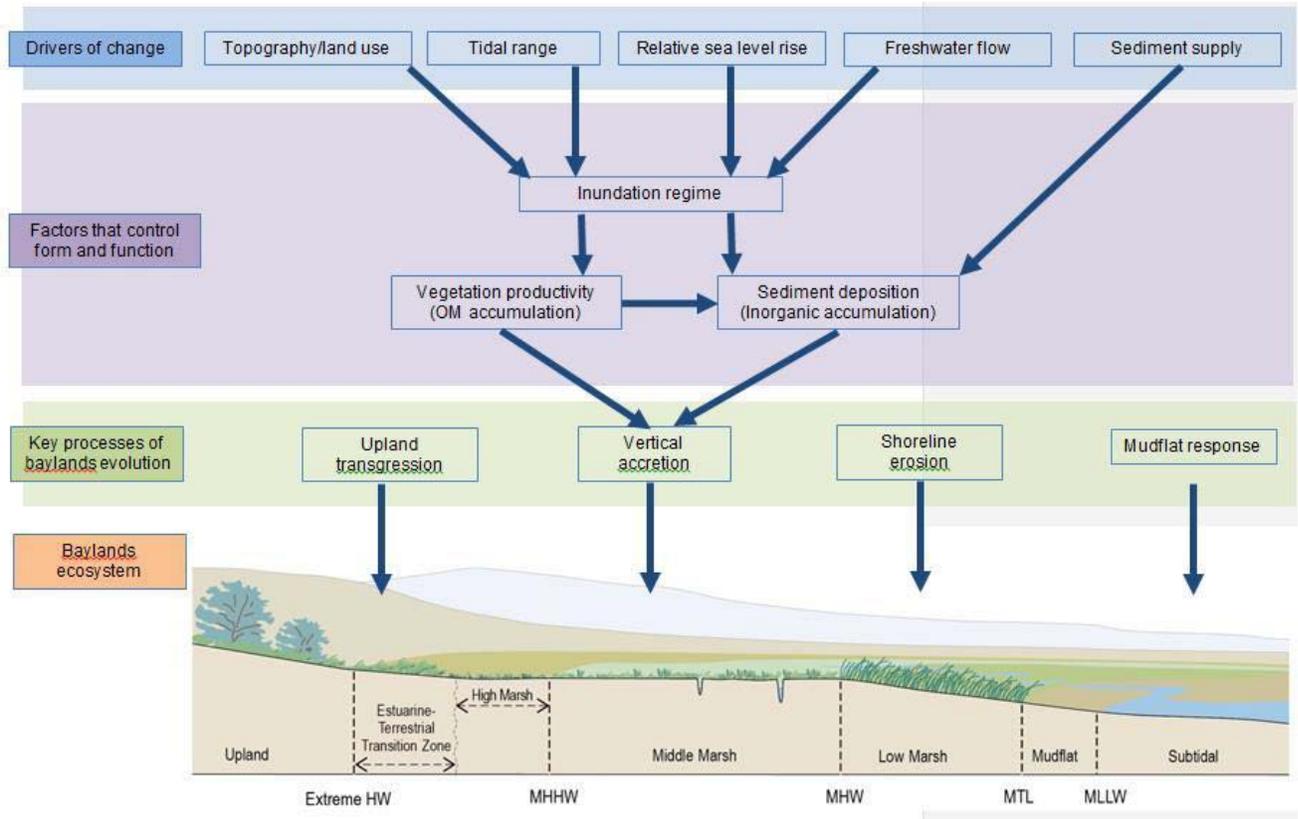
## **DROWNING**

Historic data indicates that tidal marshes in San Francisco Bay can withstand greater rates of sea-level rise than currently exist (2-3 mm/year, NOAA 2005), as long as suspended sediment concentrations remain relatively high and other factors, such as subsidence, remain relatively constant. For example, many low marsh stations within San Francisco Bay tidal marshes have accumulation rates of 6 mm/year or more. Data from other regions in the country that have experienced marsh loss give some help in identifying threshold rates of sea-level rise that may lead to tidal marshes losing elevation. For instance, in both Chesapeake Bay and Louisiana, substantial wetland loss has occurred in regions with high rates of local subsidence, with few areas able to keep pace with rates of sea-level rise greater than 10 to 12 mm/year. Thresholds for a particular location will also be affected by suspended sediment concentrations, wave regimes, and other factors. Very few sites can maintain elevation if local rates of sea-level rise are close to these values, unless suspended sediment concentrations are very high. In addition, a number of modeling studies indicate that thresholds for maintaining marsh elevation in the face of increased sea-level rise are likely in this range (see Science Foundation Chapter 2).

Increases in rate of sea-level rise will lead to changes in sediment processes within tidal wetlands (Figure 1.5). An increased rate of sea-level rise will lead to a higher frequency, greater depth, and greater duration of tidal flooding. Both mineral and organic matter accumulation will respond to this increase. Increased tidal flooding will likely increase rates of mineral sediment deposition because of longer time periods for suspended sediment to drop out of the water column, as well as greater inputs of suspended sediment with a deeper water column. However, greater water depths could lead to greater wave effects, effectively reducing sediment accumulation and potentially promoting sediment erosion. As indicated above, any local changes in suspended sediment supply would also directly affect the rate of sediment accretion at a particular location.



**Figure 1.4.** Conceptual diagram of marsh scarp evolution for prograding marsh (Beagle et al. (2015), adapted for San Pablo Bay from Allen 1989).



**Figure 1.5.** Conceptual box and arrow model showing major drivers of vertical accretion.

Within a particular marsh, there are also predictable spatial patterns in mineral sediment deposition, with greater rates of accumulation closer to tidal breaches, as well as bordering tidal channels (and leading to creation of slightly elevated natural levees along tidal creeks). For example, accretion rates at stations within the restored Island Ponds (Pond A21) in the Far South Bay were highly correlated with distance from tidal breach (Callaway et al. unpublished data). Similar results have been found for the effect of distance from a sediment source on mineral accumulation rates within natural salt marshes. As marshes are flooded more, there will likely be some positive feedback to maintain elevation, as lower elevations will lead to greater rates of mineral sediment inputs. However, this feedback depends a lot on the concentration of available suspended sediment.

Similarly there is a strong feedback between the inundation regime (frequency, depth and duration of flooding) and organic matter accumulation rates within the marsh. At very low elevations within the marsh, primary production is inhibited by increased stress from anaerobic conditions associated with high rates of inundation. At the upper end of the marsh, salt stress (and potentially competition with non-wetland species) leads to a reduction in wetland primary productivity. Together, these two factors typically result in a peak of productivity somewhere close to or just below marsh plain elevations (Morris et al. 2002). Schile (2012) found support for this these critical elevation bands favored by specific plant species across tidal marshes in San Francisco Bay, and it also has been shown to occur in a number of other tidal marsh ecosystems.

As a result, a slight increase in inundation rates could lead to an increase in plant productivity, as long as the initial plant community is at a starting elevation above the elevation of peak productivity. If inundation

is increased at initial lower elevations (or if cumulative increases in tidal flooding are great enough over time to push the plant community over this threshold elevation), it will result in reduced plant productivity. Plant productivity reduction will lead to a positive feedback that causes further loss of elevation and increased inundation because reduced accumulation in organic matter will not offset the increased rate of inundation.

If a marsh does begin to lose elevation, it may take substantial time for a marsh that is at a high starting elevation (i.e., has substantial elevation capital (Cahoon and Guntenspergen 2010) to lose enough elevation to get to the critical point at which marsh production begins to be reduced. However, if this point is reached, the marsh is likely to lose elevation even more quickly as organic matter productivity is reduced. Eventually the marsh will reach elevations that are so low that no tidal marsh vegetation can continue to grow.

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## ELEVATION CAPITAL

Elevation capital is determined in large part by comparing the absolute elevation of a marsh with the local water levels and tide range (Cahoon and Guntenspergen, 2010). Swanson et al (2013) presents a dimensionless indicator ( $z^*$ ) of elevation capital based on mean sea level and tide range:

$$z^* = \frac{z - MSL}{MHHW - MSL}$$

This non dimensional parameter is simple to calculate using existing data (marsh elevation, e.g., from LiDAR and a nearby tidal datum) and makes it possible to compare marshes with different elevations and tide regimes. Figure 1.6 shows an example of  $z^*$  for a portion of San Pablo Bay in San Francisco Bay. Many of the remaining tidal marshes have a high elevation capital ( $z^* > 1$ ), while some diked and subsided areas further inland lie well below the tide range ( $z^* < 1$ ).

As sea-level rises, wetlands can increase elevation capital by growing vertically through inorganic sedimentation and organic accretion. Inorganic sedimentation occurs when high tides inundate the marsh and suspended sediments settle out onto the marsh. As sea-level rises, the marsh is inundated more frequently, which increases the amount of sediment available for deposition. Organic accretion occurs through accumulation of organic matter from above- and below-ground plant growth. Morris (2002) provides one of the first measurements of aboveground productivity in a salt marsh from South Carolina, where organic accretion plays a large part in building elevation capital. Figure 1.7 shows the Morris 2002 results in terms of  $z^*$  values. In this case productivity peaked at approximately 40 cm below Mean High Tide ( $z^* = \sim 0.3$ ). Below this elevation (at higher levels/frequency of inundation) the system becomes unstable and plants begin to drown.

Swanson et al (2013) fitted similar curves to four marshes in San Francisco Bay (also shown in Figure 1.7) and found that peak organic accretion occurred at a lower rate and at higher  $z^*$  values than the South Carolina site. This study also estimated inorganic sedimentation at various water levels for the four San Francisco Bay sites, shown in Figure 1.8.

Organic and inorganic accretion must be considered together because each process is a function of a marsh's elevation relative to the tide range and therefore each influences the other. Figure 1.9 shows the

two processes added together for the San Francisco Bay sites. It is notable that this figure does not differ significantly from Figure 1.8 because inorganic sedimentation dominates at these sites and so sediment supply is an important driver for accretion rates. The maximum rate of inorganic sedimentation is between 15 and 60 times greater than the maximum rate of organic accumulation. Inorganic sedimentation will therefore be the primary process for San Francisco Bay's marshes to build elevation capital with increasing sea levels. Fortunately, inorganic sedimentation increases with decreasing elevation and does not exhibit the same instability that organic accretion experiences at low elevations. However, as the rate of sea-level rise increases and sediment supply decreases, the maximum rates of accretion/sedimentation may eventually be unable to maintain elevation capital.

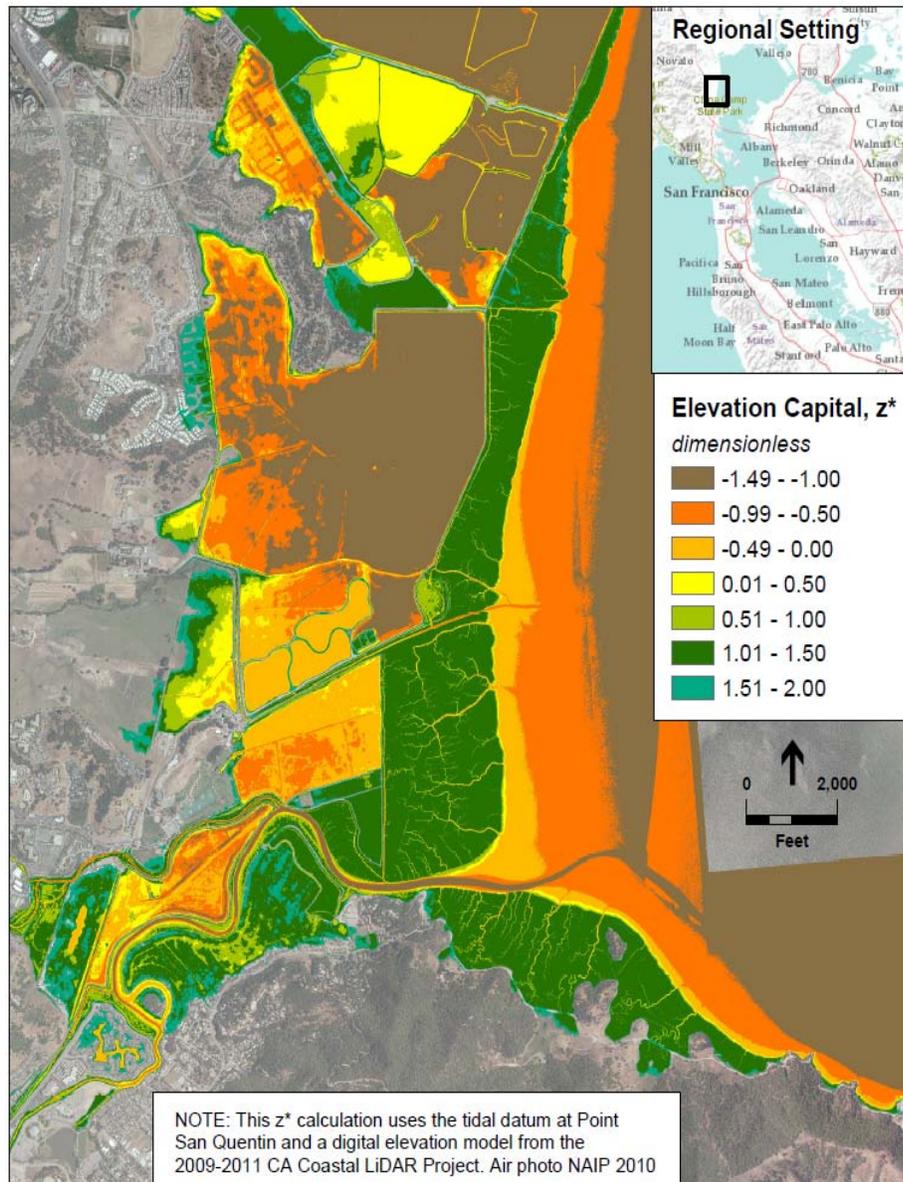
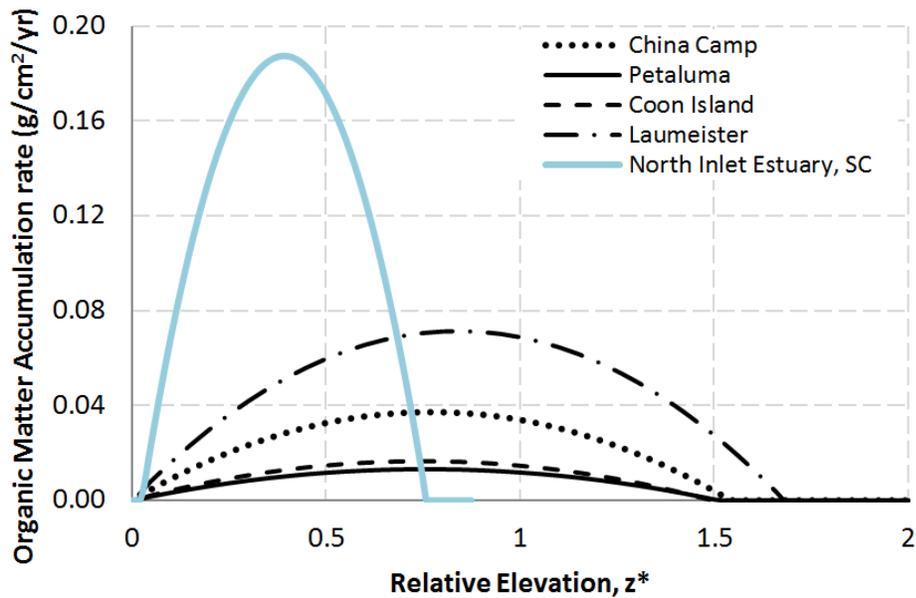


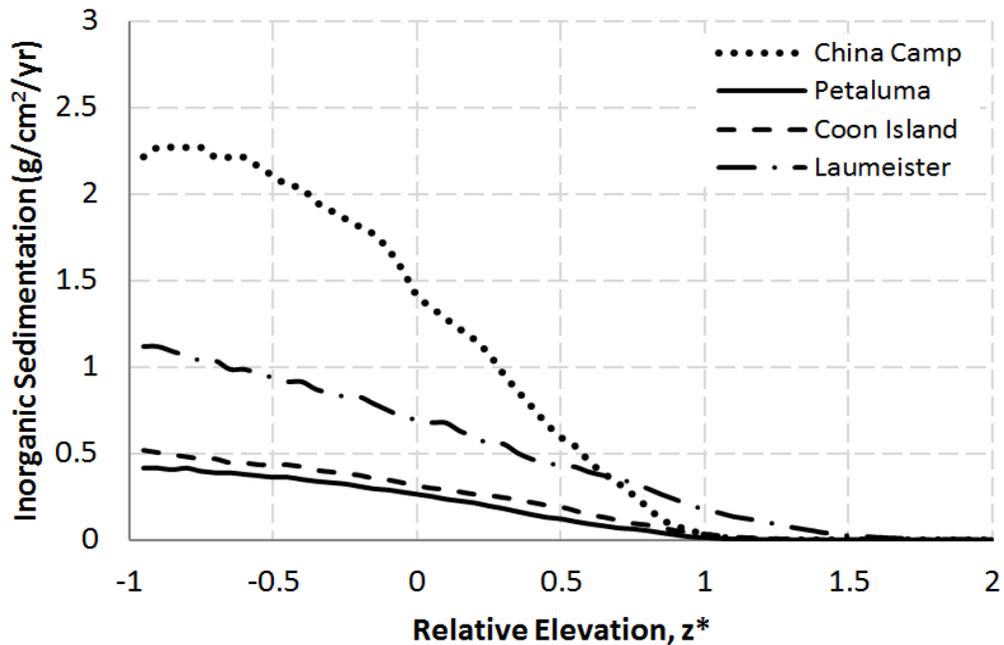
Figure 1.6. Elevation capital ( $z^*$ ) at China Camp and neighboring marshes.

### Organic Matter Accumulation



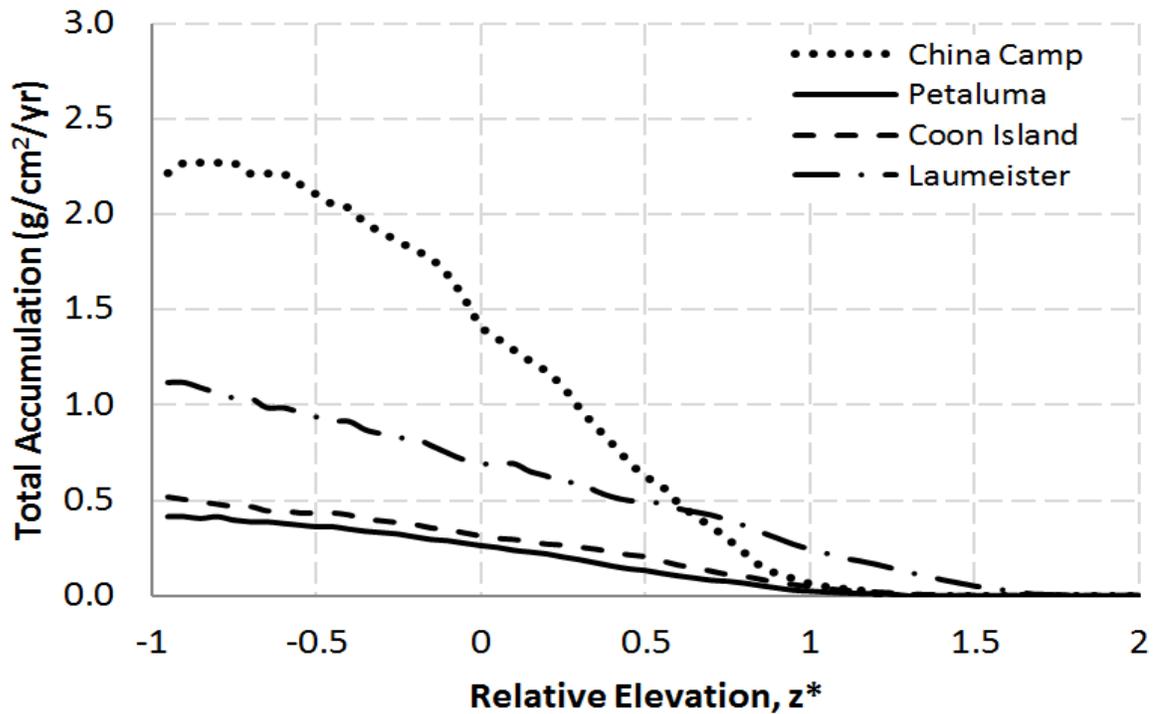
**Figure 1.7.** Salt marsh productivity at various water levels. Organic matter accumulation increases with elevation. Above the peak productivity, marshes are stable because as sea-level rises, organic accumulation will increase (up to a certain threshold). Below the peak productivity marshes are unstable (as water levels rise, salt marsh productivity will go down). Derived from Morris 2002 and Swanson et al 2013.

### Inorganic Sedimentation



**Figure 1.8.** Inorganic sedimentation for varying elevations relative to the tides. Inorganic sedimentation is higher for marshes lower in the tide range. Derived from Swanson 2013.

## Total Accumulation



**Figure 1.9.** Total accumulation (inorganic sedimentation and organic accretion) for four sites in San Francisco Bay. Derived from Swanson et al 2013.

### HOW HAVE THE BAYLANDS EVOLVED IN THE PAST?

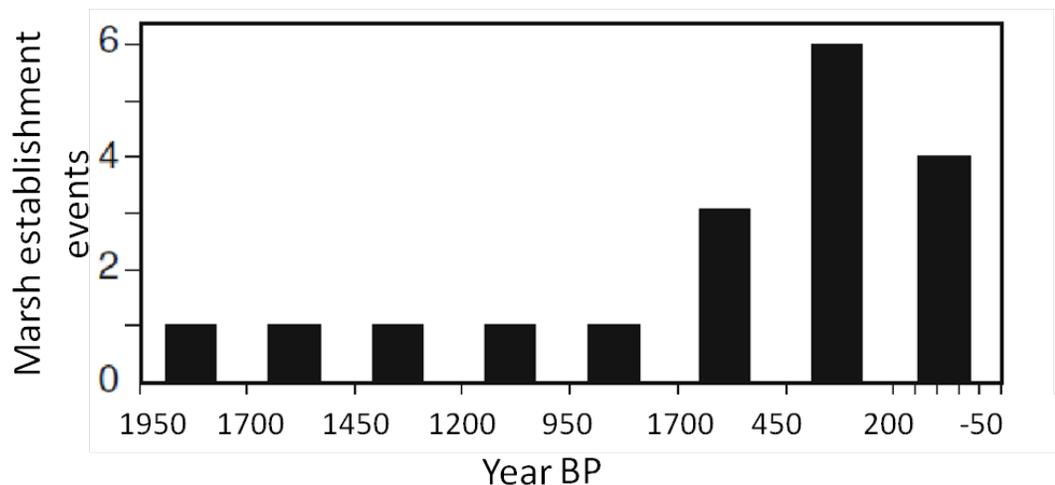
The Bay’s evolution during the historical and late Holocene periods suggest the potential for resilience to climatic variation as well as the challenges and opportunities for enhancing that adaptive capacity. This section looks to the past to show how the Baylands have adapted to significant environmental stressors and how they might do in the future. The Goals Project (1999) briefly considered Holocene geomorphic evolution of the estuary, and its historical ecology, in relation to future climate change as context for the formulation of regional and subregional goals:

“Between 2000 and 3000 years ago, mudflats and tidal marshes began to form around the edges of [prehistoric baylands]...Some of the current global climate change models predict future rates of sea-level rise that exceed the early rates for the Estuary (Gleick et al. 1999). How the baylands might respond to such a rapid increase in sea level is unknown. Their response will depend on the supplies of sediment and runoff, which may increase or decrease with climate change, depending partly on how the land is managed.” (Goals Report 1999).

The Goals Project (1999) greatly advanced the understanding of pre-modification Baylands habitats, their ecological functions, and associated physical drivers as to inform the setting of habitat goals (Goals Report 1999, Gedan et al., in press).

Scientific understanding of late Holocene evolution of the Estuary's tidal marshes, especially in relation to paleoclimate and rates of Holocene sea-level rise, has continued to develop in the decade following the Goals Project (Malamud-Roam et al. 2007). This provides an opportunity to update regional wetland ecosystem goals and also provide evidence of long-term ecosystem responses to similar environmental drivers that face us in the future. These include higher rates of sea-level rise, limited sediment supply, warmer and drier climates, and extreme flood events.

Using cores, Atwater et al. (1977, 1979) established robust estimates of millennial-scale marsh accretion rates under late Holocene sea-level rise, and accurate ages of tidal salt marshes. They found that the tidal marsh bayland landscape began developing only 2000-3000 years B.P., and only after rates of sea level slowed to 1-2 mm/yr. This has been recently confirmed by analysis of cores and sedimentation modeling in the South Bay, which shows that marshes in the central portion of the southern San Francisco Bay dated to 500 to 1500 years BP, while expansion of marshes in southernmost San Francisco Bay dated to 200 to 700 years BP (Figure 1.10).



**Figure 1.10.** South San Francisco Bay marsh establishment for 250-year periods over the last 2000 years. From Watson and Byrne (2013).

The USGS San Francisco Bay cores also show that tidal marsh plains either did not develop or were eroded in the early-mid Holocene epoch. Estuarine erosion rates were as rapid as 30 m/year during this period (a combination of horizontal shoreline erosion and submergence of alluvial plains) and sea-level rise rates reached an average of about 20 mm/year or about 10 times the 20th century rate (Atwater et al. 1977). The estuary was characterized by migrating fringing or deltaic marshes (on the time scale of centuries) rather than stable tidal marsh platforms. Drainage of these marshes was somewhat different than mature tidal marsh and creek systems appearing later as they were delayed until sea-level rise rates were less than 1-2 mm/year. Nascent San Francisco bay tidal marshes at low but rapidly rising sea levels 8000 years BP did not extend farther inland than Hunters Point, San Francisco, and they were brackish, due to relatively small tidal prism, and grew more saline as the Bay submerged and enlarged (Atwater et al. 1977).

The earliest stages of San Francisco Bay historical salt marsh development indicate instability as sea-level rise gradually slowed. Basal marsh sediment sequences alternated with lower intertidal and subtidal bay mud, indicating that fringing marshes were drowned by rising sea levels and then subsequently re-formed (Atwater et al. 1977). More recent high-resolution cores of prehistoric remnant marshes at China Camp

(Goman et al. 2008) and Rush Ranch (Byrne et al. 2001) are also consistent with repeated reversion between marsh and mudflat, perhaps due to accelerations of sea-level rise or erosional processes.

Analysis of stable carbon isotopes, pollen, plant macrofossils, and other salinity proxy data in cores reveals that Suisun and San Pablo Bay marshes have alternated between brackish and salt marsh vegetation over multiple centuries of warm and dry or cooler and wet climate periods (Malamud-Roam et al. 2007). Centuries of higher-than-average salinity in the Estuary have been associated with reduced fresh water inflows during 1600–1300 years BP, 1000–800 years BP, 300–200 years BP, and from about AD 1950 to present. Periods of lower-than-average salinity due to increased fresh water inflow occurred before 2000 years BP, from 1300 to 1200 years BP and from about 150 years BP. to AD 1950 (Malamud-Roam and Ingram 2004). These records suggest that California's climate since A.D. 1850 has been unusually stable and benign compared to climate variations during the previous 2,000 or more years.

Paleoecological investigations of the Estuary's tidal marshes have also contributed to a greater understanding of marsh dynamics in response to climatic variability. Brackish tidal marshes at Rush Ranch in Suisun Marsh originated as salt marshes alternating with mudflats only about 2500 years ago, and they did not evolve into a stable equilibrium state of brackish tidal marsh. Tidal marshes in Suisun Bay alternated between phases of high salinity and low salinity lasting several centuries (Byrne et al. 2001), consistent with paleosalinity history of other marshes in Suisun Bay and Carquinez Straits (Malamud Roam and Ingram 2004). Evidence of repeated extreme storm events was also detected in some marsh cores, such as sand layers (consistent with extreme slopewash and alluvial fan deposition) deep into tidal marshes at China Camp (Goman et al. 2008). This suggests that the Estuary's tidal marshes have been responding to wide swings of climate and extreme meteorological events during their brief 2000-3000 year history. While the Estuary's marshes have exhibited dynamic responses in vegetation characteristics, morphology, and extend during the late Holocene, they have also exhibited substantial persistence through climatic variations, presumably sustaining native species through the shifts. This is likely possible because of factors that conferred resilience on the baylands ecosystems, including habitat connectivity, uninterrupted sediment supply, and adjacent transition zone migration space.

Modern biological conservation implications of the Estuary's recent paleoecological investigations are just beginning to be applied to climate change adaptation planning (Grewell et al. in press). The late formation of mature channel systems and prolonged fluctuations between brackish and salt marsh salinity regimes in the northern Estuary during the late Holocene, suggest that species distributions associated with past marsh salinity gradients and habitat configurations have either adapted or moved across whole subregions of the Estuary, and they persisted for centuries before abruptly switching with climate shifts.

Paleoecological studies of nearby West Marin marine lagoons, such as Bolinas Lagoon, provide additional insights into how tidal marsh assemblages similar to those of the Estuary cope with much higher rates of relative sea-level rise. For example, instantaneous submergence events of 45 cm associated with co-seismic subsidence of San Andreas fault activity recur about every few centuries, in addition to sea-level rise, causing higher rates of relative sea-level rise (Byrne and Reidy 2005). Interestingly, tidal marshes of Bolinas Lagoon persist or regenerate in confined reaches of the lagoon where deltas are deposited by fluvial and tidal processes (Byrne and Reidy 2005) and this could be a guide for more resilient shoreline types in the Bay. Other subsided San Andreas Fault lagoons with high relative sea-level rise (Bodega Harbor, Tomales Bay; Niemi and Hall 1995; Grovel et al. 1995) also exhibit coupling of tidal marsh development with fluvial delta sedimentation, consistent with global coastal geomorphic models of tidal marsh development in submerging lagoons (Woodroffe 2002, Cooper 1994).

## DRIVERS OF CHANGE

This section reviews existing drivers of change and how they are likely to influence the evolution of the future baylands landscape. These drivers include sediment supply, freshwater inflows, and salinity coupled with new drivers due to climate change, namely sea-level rise, as well as temperature and precipitation.

### Sea-level rise

The sea-level rise projections for the West Coast have recently been provided by the National Academy of Science National Research Council study (NRC, 2012). For San Francisco Bay, NRC (2012) project a regional sea-level rise, including an allowance for vertical land motion, of between 5 to 24 inches by 2050 with a mid-projection of 12 inches and 17 to 66 inches by 2100 with a mid-projection of 36 inches (Table 1.1).

**Table 2.1.** San Francisco Regional Sea-level rise Projections Relative to Year 2000 (NRC 2012, OPC 2013)

Year	Projection (A1B scenario)	Range (B1 and A1F1 scenario)
2030	6 inches (14 cm)	2-12 inches (4-30 cm)
2050	12 inches (28 cm)	5-24 inches (12 to 61 cm)
2100	36 inches (92 cm)	17-66 inches (42-167 cm)

The NRC values include subsidence of  $1.5 \text{ mm yr}^{-1}$  for all of California south of Cape Mendocino due to deep tectonic movements. This is a rough estimate that does not take into account localized variations in vertical land motion due to shallow subsidence and local tectonic movement. Observations of vertical land motion more specific to the South Bay (Burgmann, 2006; Gill, 2011; USGS 2012) are inconsistent in direction, and ranged from 1 mm/yr of subsidence (USGS 2012) to 1.5 mm/yr of uplift (Burgmann, 2006). For consistency with the NRC approach and to provide a slightly more conservative (higher) estimate of sea-level rise, we use the NRC assumption of vertical land motion for this study.

The selection of sea-level rise scenarios depends on the planning horizon of the study. Projects implemented during the planning timeframe must be built not just for sea-level rise at the time of construction, but for sea-level rise occurring during the life of the restoration. At the same time, projection too far forward in time becomes increasingly uncertain and may be better accommodated by providing adaptive capacity in the design of the project:

*“Until 2050, there is strong agreement among the various climate models for the amount of SEA-LEVEL RISE that is likely to occur. After mid-century, projections of SLR become more uncertain; SLR projections vary with future projections due in part to modeling uncertainties, but primarily due to uncertainties about future global greenhouse gas emissions, and uncertainties associated with the modeling of land ice melting rates. Therefore, for projects with timeframes beyond 2050, it is especially important to consider adaptive capacity, impacts, and risk tolerance to guide decisions of whether to use the low or high end of the ranges presented.”*

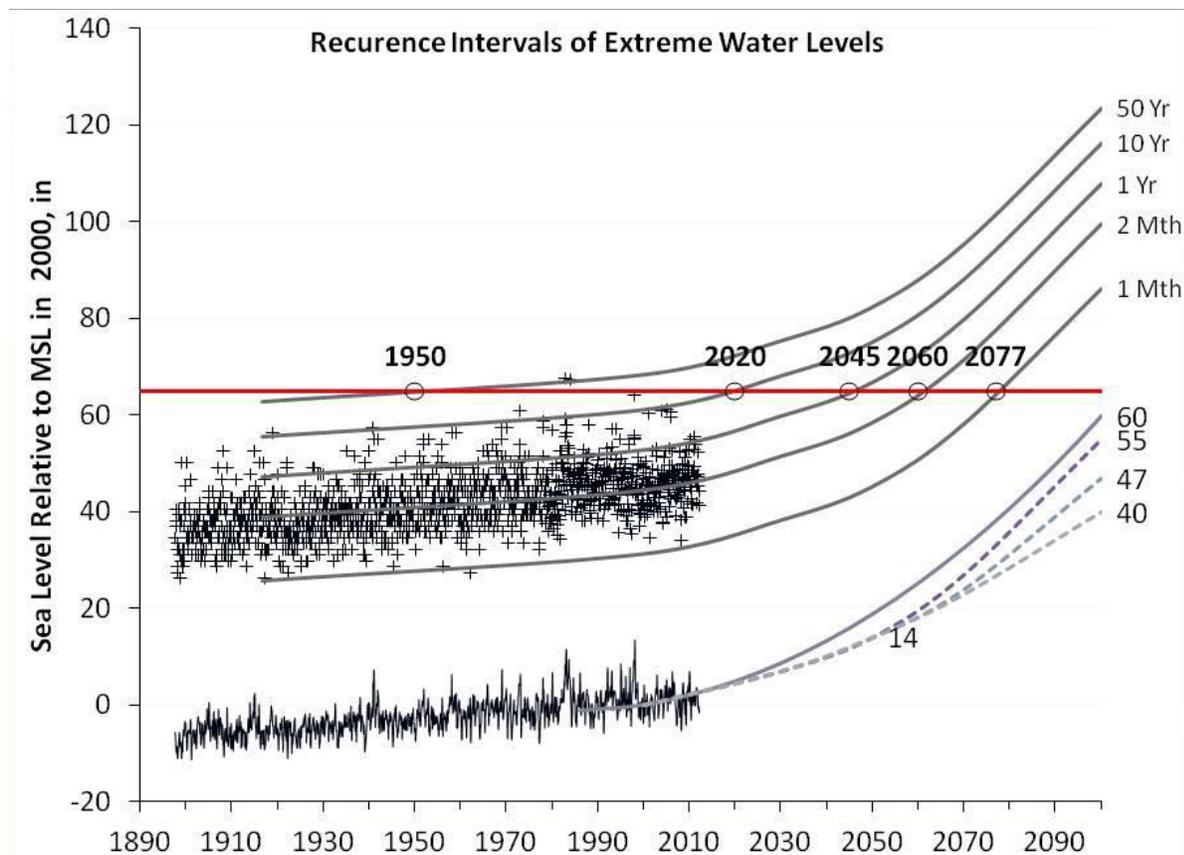
page 3, OPC (2013)

The projections of sea-level rise relative to the year 2000 used in this study, is consistent with NRC (2012), OPC (2013) and the South San Francisco Bay Shoreline Study, are:

- 12 inches by 2050
- 36 inches by 2100

### Extreme Water Levels and Recurrence Intervals

The above discussion centers on mean sea level. However, the first impacts of sea-level rise that will affect marshes will be from extreme events as shown in Figure 1.11 below. Storm conditions represent a lower frequency event, they come with a larger potential flooded area with deeper flooded depths, higher velocities, and a greater likelihood of wind driven waves that could increase erosion. Figure 1.11 shows that as mean sea level raises so will the elevation events of a fixed recurrence. For a fixed elevation the frequency of being inundated will increase over time and inundation depths will increase. In the example shown in Figure 1.11, a marsh inundated with a 10 year return interval in 2020 will become a 1 year return interval by 2045. For marshes, extreme events will have a much earlier impact before mean sea level. The inundation regime will be changed well before the site is permanently inundated by mean sea level.



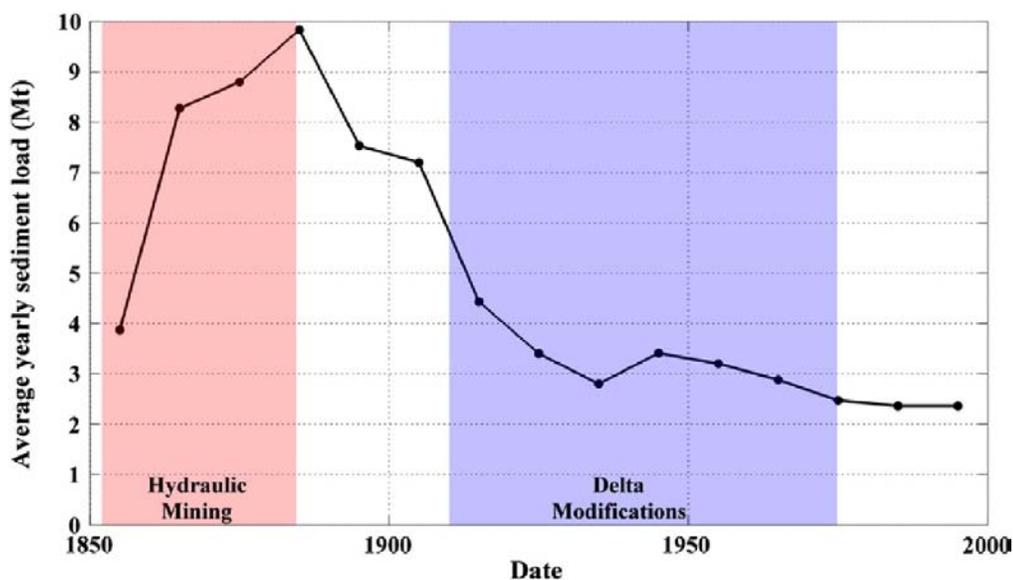
**Figure 1.11.** Recurrence Intervals of Extreme Water Levels in San Francisco Bay. NOTE: Historical (solid black line) and annual extreme water levels (black crosses) from Presidio tide gauge. Dotted lines indicate NRC 2012 projections. Year 2000 recurrence intervals from ESA PWA (2011). Developed from an idea of David Kriebel, USNA.

## Sediment Supply

The Goals Report (1999) emphasized the role of fine sediment (bay mud) depositional processes in development and maintenance of tidal marsh and mudflat habitats within the Baylands ecosystem. Coarse sediment – sand, shell, and gravel – was also identified as the substrate of historical and modern estuarine beach habitats within intertidal and supratidal zones of the Estuary. Sediment supply to the Bay has seen considerable variation over 170 years in common with many other estuary systems. Disruptions in Bay watersheds increase sediment loads, followed by dams, water diversions and altered river management that reduce variability and thus sediment supply (Barnard et al., 2013). Future availability of both fine and coarse sediment in the Estuary is likely to continue to change and will likely impact evolution of existing beaches and wetlands and also constrain the adaptation measures that might be considered in response to climate change.

Extensive hydraulic mining in the Sierras, coinciding with a period of abnormally high regional precipitation mobilized large volumes of sediment to San Francisco Bay during the late 1800s (Barnard et al 2013). This led to significant changes to Bay bathymetry, beaches, and fringing tidal marshes: a comparison of 1856 and 1887 bathymetric surveys of San Pablo Bay by Jaffe et al. (2007) shows that the estuary accumulated sediments during this period, with intertidal mudflats expanding by 60%. Efforts in the early to mid-1900s to manage floods, develop hydropower, and deliver water supplies led to the construction of dams trapping sediment throughout the Sierra Nevada which, together with cessation of mining in 1884, cut off the supply of hydraulically-mined sediment to the Estuary (Schoellhamer, 2011; Wright and Schoellhamer, 2004). The main pulse of bed sediment passed Sacramento by 1950 (Meade 1950).

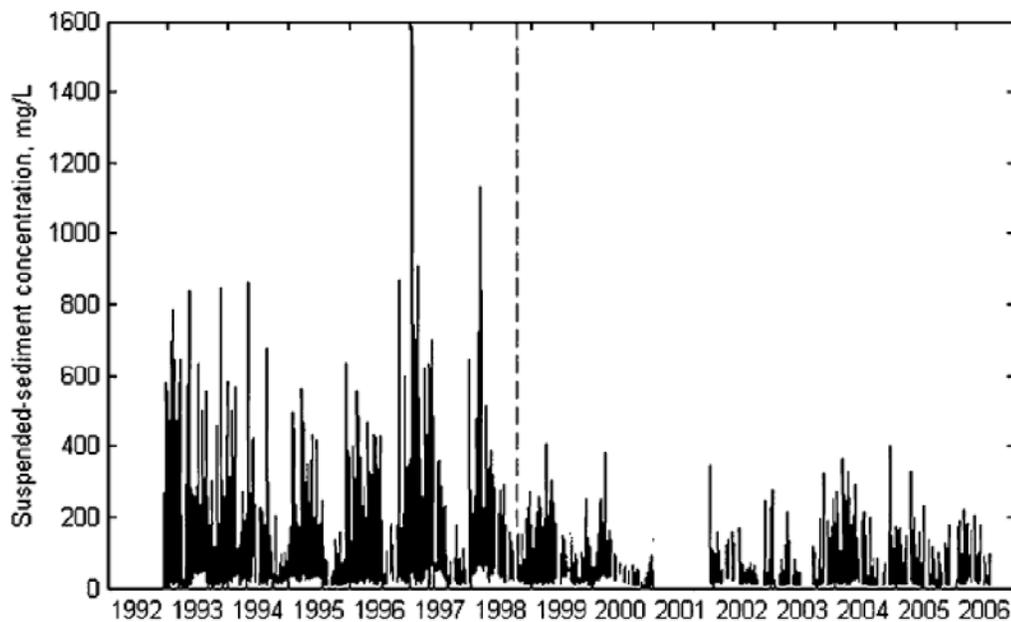
Simultaneously, levee construction isolating floodplains from the main rivers together with logging, urbanization, agriculture and grazing activities increased sediment yields from local watersheds in the mid to late 20th century (Lewicki and McKee, 2010). As urban areas matured and erosion rates stabilized at the end of the 20th century so sediment yields have decreased in a number of watersheds (McKee et al 2004). Ganju et al (2008) estimated decadal sediment input from the Delta as shown in Figure 1.12.



**Figure 1.12.** Decadal sediment load from Sacramento-San Joaquin Delta (from Ganju et al. 2008) showing periods of hydraulic mining and Delta modification. Source: Barnard et al. 2013.

At least 200 Mm<sup>3</sup> of sediment has been permanently removed from the Bay by navigation dredging, aggregate mining and borrow pit mining (Dallas and Barnard 2011). Sediment was also taken out of the system by filling and diking 90% of the Bay's tidal wetlands, trapping the sediment behind levees making it unavailable to mudflats and marshes (Atwater et al. 1979).

Reduction in supply and continual removal of sediment from the Bay has impacted suspended sediment concentrations (SSC) in a measurable way. Schoellhamer (2011) shows observations of near-surface and mid-depth SSC at most deep channel sites in the Bay from the early 1990s to water year (WY) 1998 were almost double that from WYs 1999 to 2007; the only exception was San Mateo Bridge. Figure 1.13 shows such a step change for observations made in San Pablo Bay. In general there was a statistically significant 36% decrease in SSC in San Francisco Bay from water years 1991–1998 to 1999–2007.

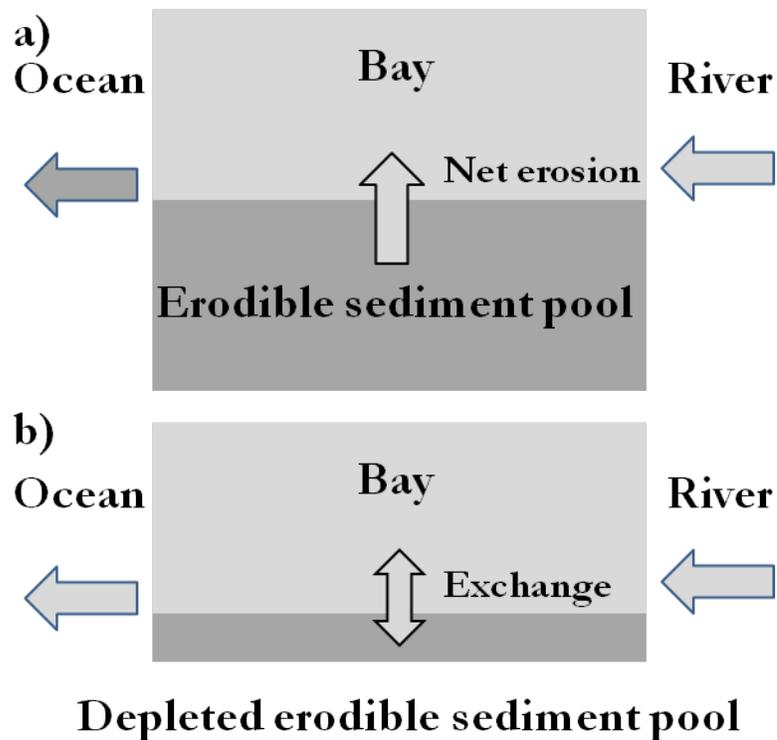


**Figure 1.13.** Suspended sediment concentration, mid-depth, Point San Pablo from 1993 to 2006. The vertical dashed line indicates when the step decrease occurred. Source: Schoellhamer 2011.

These steps cannot be explained by variations in fluvial sediment supply, rather Schoellhamer (2011) provides a hypothesis that the Bay had an erodible pool of sediment that was depleted in the late 1990s and the Bay switched from a transport-regulated regime to a supply-regulated regime. A quantitative conceptual model demonstrates that crossing the threshold between regimes can rapidly reduce suspended sediment mass and lead to rapid clearing of the water column (Figure 1.14).

Prior to the step decrease, Bay SSC would remain high in water years with little watershed sediment supply because the erodible sediment pool supplied suspended sediment and SSC was transport-regulated. Schoellhamer (2011) estimates the erodible pool in the mid-1900s was about 60 times the volume of the mean annual sediment supply. As sediment supply diminished over the 20th century, so the erodible pool was depleted although it was still transport-regulated. When the erodible pool becomes sufficiently depleted, however, its buffering action is lost. Then the suspended sediment becomes supply-regulated and much more sensitive to the supply from the local watersheds, increasing their importance and the

desirability to reconnect them with the tidal marshes. This appears to have occurred in 1999 in San Francisco Bay.



**Figure 1.14.** Conceptual model of an erodible sediment pool that becomes depleted (a), reducing suspended sediment concentration (b). Source: Schoellhamer 2011.

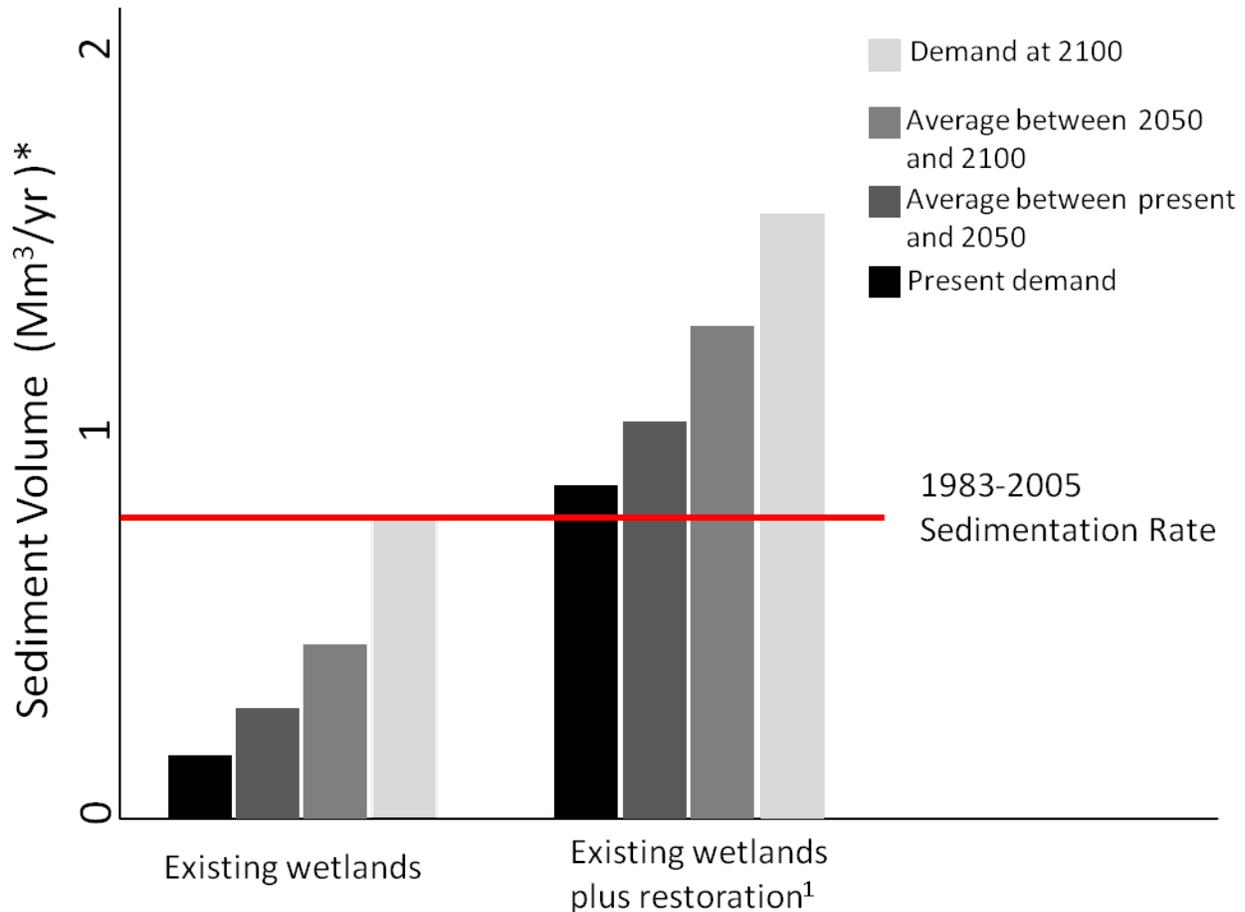
Looking ahead and noting the size of the erodible pool in the last century it is likely that, even in wet years (e.g., 2006), rivers will not be able to supply enough sediment to restore the pool. Thus, SSC is likely to remain low in the future. As SSC becomes more closely linked to the supply during individual water years, it will also reflect decreases in the river supply. Hestir (2004) found that river supply to the estuary decreased about 1.3% per year between 1975 and 2008, so future reductions in SSC should be anticipated.

There is a high probability that the sediment budget is likely to shift toward deficit because of sediment demand and retention in old and restored or recreated marshes as well as inadvertently flooded Delta islands. The sediment supply to the estuary may decrease further, particularly if more instream storage is built, although that trend is uncertain (McKee et al. 2006, Cloern et al. 2011).

### Sediment Demand

Jaffe et al. (2011) made preliminary estimates of the volume of sediment needed per year to maintain tidal flats, tidal marshes, marsh restoration, and for the bay floor to keep up with projected future sea-level rise. Figure 1.15 depicts scenarios for the present sediment demand, from the present to 2050, demand averaged from 2050 to 2100, and demand projected for the year 2100. Jaffe et al. (2011) suggest that without marsh restoration efforts there may be enough sediment in the South Bay to keep pace with sea-level rise. With the projected additional demand for sediment under various marsh restoration scenarios, sediment supply

may also not be able to keep pace with the demand. These projections just evaluate inorganic sediment contribution to marsh maintenance. The organic contribution (life plant growth) to marsh accretion was not considered. Organic contribution to marsh sediments in San Francisco Bay salt marshes is between 7 – 15% (Watson 2004, Callaway et al. 2012).



**Figure 1.15.** Projected sediment demand from tidal flats and marsh restoration under future sea-level rise scenarios (Mm<sup>3</sup> = million cubic meters). \*Does not include organic contribution to marsh accretion. <sup>1</sup>35 Mm<sup>3</sup> over 50 years = 0.7 Mm<sup>3</sup>/yr (Shoellhamer et al. 2006).

The basic assumption is that processes and inputs that control tidal flat and marsh maintenance will not change in the future. Although the basic processes (e.g., wind wave suspension) are not likely to change, their effectiveness may change with sea-level rise. For example, if future sea-level rise does not result in bed sediment accumulation (and therefore an increase in vertical elevation), there will be larger waves, which could result in more erosion of the mudflat and therefore less mudflat. The main controlling factor is sediment supply (from tributaries, and erosion of tidal flats north of Dumbarton Bridge): if it decreases, particularly with the loss of the erodible pool of sediment, and switches to supply-limited regime postulated by Schoellhamer (2011), then sedimentation rate decreases, and mudflats extent decreases.

### Topography

Topography is key not only to the location of tidal habitats but also to its quality. Within-marsh complexity has been lost in the last 40 years during restoration as restored marshes have not established for a long

enough period for complex channels and topographic features to develop. New marsh restoration techniques in recent years have been more focused on creating topography within the marsh. However as sea-level rises the upland topography adjacent to the marshes will play an increasingly important role in facilitating or preventing the landward migration of marshes into the transition zone (transition zones are described in detail in Science Foundation Chapter 4). Transition zones provide much of the land area that will serve as accommodation space as sea-level rises. Accommodation space provides area for marshes to migrate landward with increasing sea levels as new tidal marsh is created at the margin of existing marshes. The width of this new marsh is controlled by the topographic slope along the margin of the bayshore, in addition to other factors such as rates of sea-level rise, suspended sediment concentrations, freshwater inputs, and organic matter inputs.

Most of the historic transition zones around the baylands have been diked and leveed, partly filled, and are part of the built environment. Where available land exists, there are competing interests for preservation and development. Transition zones will be squeezed between a rising bay and human developments as the seas rise. Developed areas may in turn provide accommodation space in the future if they are abandoned due to rapidly rising sea levels.

In the remaining areas where land is still available, these transition zones are narrower where the tidal range is lesser and the land is steeper. Narrow transition zones will be short-lived with increasing water levels. Where the tidal range is greater and the land slopes more steeply to the bayshore, the transition zone is wider provided there is no development in the way. Broad areas where migration can accommodate the services of the transition zone can increase the landward extent of the transition zone, particularly in areas that are not steep. Wide gently sloping transition zones will last longer if connected to active estuarine and terrestrial processes.

Artificial levees comprise much of the existing transition zones and most of the existing levees provide very little transition zone habitat with virtually no accommodation space. Most of these existing levees cannot persist in response to increasing water levels. However, where levees are adjacent to low-gradient upland slopes there are opportunities to create accommodation space.

## Water

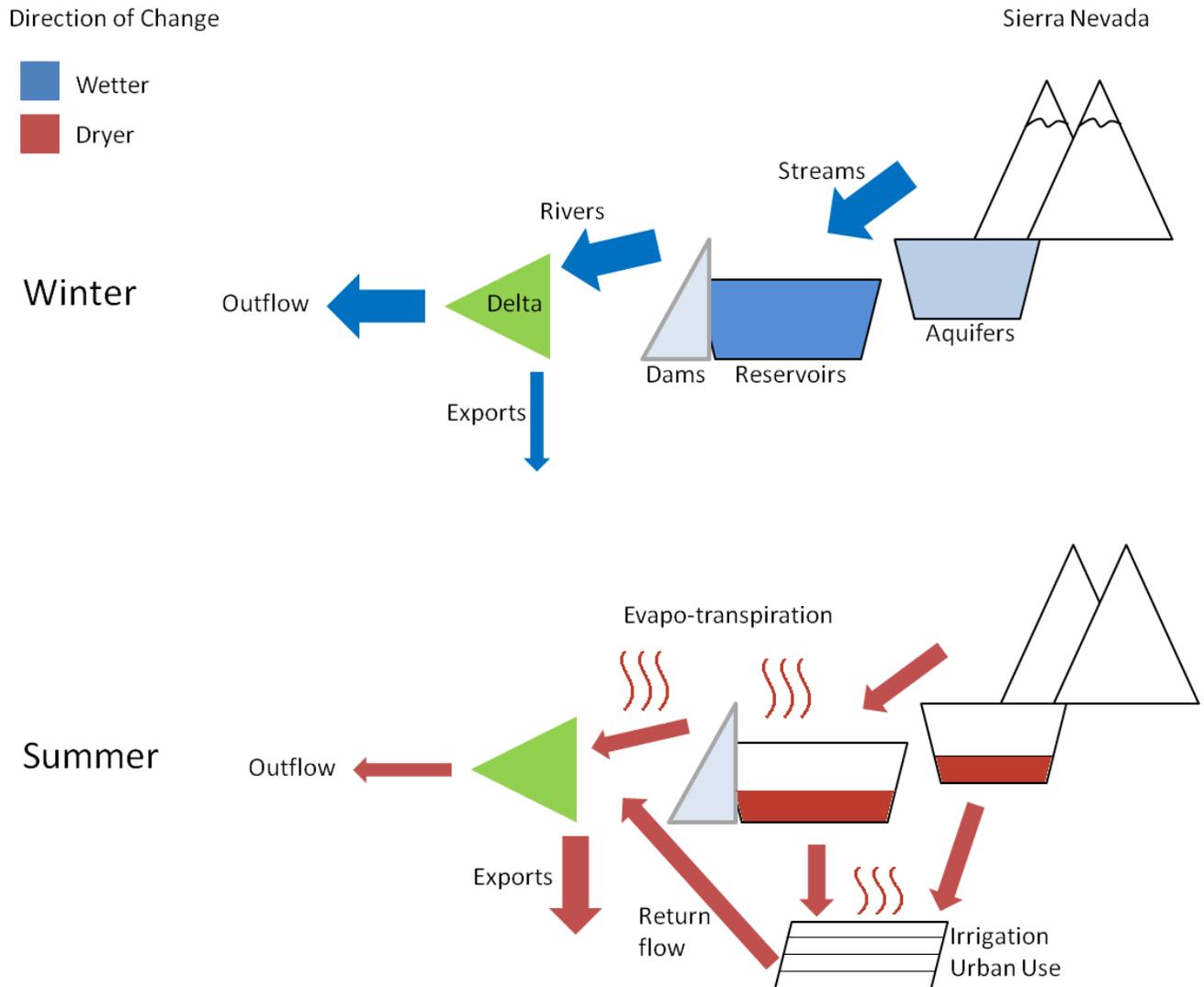
Freshwater flow into the estuary comprises on average about 99% outflow from the Delta and about 1% from wastewater treatment plants and local streams (MacWilliams et al. in prep.). Delta outflow results from quantity, timing, and location of precipitation, snowpack storage, groundwater, and reservoirs, and losses to evapotranspiration and export flows from the southern Delta. For simplicity, long-term changes in wet and dry seasons are considered separately (Figure 1.16).

Long-term changes in precipitation in the Central Valley watershed are expected through two mechanisms. Warming will reduce snowpack in high mountains, resulting earlier peak runoff (Aguado et al. 1992, Dettinger and Cayan 1995). Changes in total precipitation may occur as a result of climate change, but modeling support is weak (Dettinger 2005), and total precipitation is already highly variable. Thus the likely future scenario is for runoff that is higher in winter and lower in summer, with continued variability at all time scales.

Increasing runoff means less runoff will be captured in reservoirs and groundwater, and winter flows in the rivers and through the Delta will be higher (Figure 1.16). The likely human response will be to increase

storage capacity, which will increase variability in Delta outflow by capturing lower flows and passing through higher flows.

During the dry season, flows in high-elevation streams will be lower because of earlier snowpack loss. Water stored in reservoirs and groundwater will be used for irrigation. Industrial and residential use and wastewater will be returned to the watershed except for evapotranspiration, which will increase because of the higher temperature and the need for more irrigation water. This may be offset by improved irrigation practices and urban water conservation.



**Figure 1.16.** Schematic diagram of forecasted changes in the hydrology of the Central Valley during the wet winter season (top) and dry summer season (bottom) with an increase in air temperature. Arrows represent flows of water, with width representing relative magnitudes of changes in flow. Blue arrows represent flows or processes consistent with overall wetter conditions, and red arrows represent those consistent with dryer conditions.

Reduced inflows, greater demand, and increased evapotranspiration will reduce flow going to the Delta. There, increased demand to the south will require increased export flows from the south Delta (as now) and presumably from a new facility to be built on the Sacramento River. Some of that water will return via the San Joaquin River, some will be lost to evapotranspiration, and some will go to cities and farms outside the watershed. Additional transfer of water for urban use in San Francisco and the East Bay will also likely increase. The net result of these changes will be reduced Delta outflow.

Small streams entering the estuary are likely to see an increase in winter flows due to increased runoff from impervious surfaces as urbanization increases, and possibly also increased frequency of large storms.

Summer flows will decrease because of increased evapotranspiration in the warmer, dryer local watersheds. Flows from wastewater treatment plants and urban runoff will increase with population growth.

Changes to freshwater input are highly probable. The first observed signal of climate change effects on the estuary was the shift toward an earlier snowmelt peak (Roos 1989). This shift, caused by warming in the high Sierra, is expected to continue with warming. The outcome will be a shorter, more intense period of runoff in winter and early spring, and a protracted dry season. Thus, irrespective of any change in storm intensity (see below), the winter hydrograph will be flashier with higher flows during wet winters. During drier winters, much of the snowmelt peak may be captured in expanded reservoirs and exported in new water diversion facilities (Section 2.1E, BDCP 2013). This will likely amplify the current high interannual variability in flows through the estuary.

The mixing of freshwater with ocean water will lead to the creation of estuarine salinity gradients. The salinity distribution responds to freshwater flow, tides, sea level (affected by rising sea level in the ocean and by atmospheric pressure gradients and coastal wind), and wind within the bay. Each of these influences has a different time scale of influence on salinity. Many attributes of the estuary are in turn affected by the extent of the salinity gradient.

Freshwater flow is the predominant control of the salinity gradient on timescales of a week or longer. The position of the salinity gradient in the estuary is indexed by X2, which is the distance in kilometers up the main channel of the estuary from the Golden Gate to where the salinity is 2 ppt, roughly the center of the low-salinity zone (Figure 1.17). A large X2 value means salinity has intruded farther into the Delta. X2 is related to the freshwater discharge from the Delta with a time lag of about 2 weeks (Monismith et al. 1996, 2002).

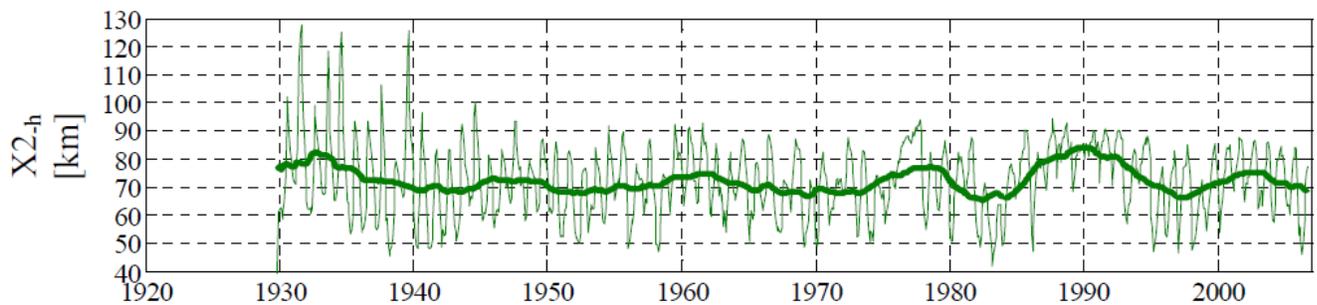


Figure 1.17. Location of X2 (km from Golden Gate) under historical flow conditions (CCWD 2010).

Tidal currents mix estuarine waters longitudinally through (a) tidal trapping, by which water masses that split into different channels, or channels and shoals, become separated from each other; and (b) by tidal pumping, by which tidal currents over irregular bathymetry induce a net circulation. An important additional influence on salinity is gravitational circulation, a density-driven circulation consisting of up-estuary tidally-averaged flow near the bed and down-estuary flow near the surface, which in turn generates vertical stratification. Strength of gravitational circulation and stratification depends on the degree of vertical mixing induced by tidal currents, which break down stratification. Frequency and strength of stratification increases with water depth, because vertical mixing affects a smaller fraction of the water column in deep water. Thus, deeper channels of the estuary can be regions of strong up-estuary salt flux, and these areas in effect limit responsiveness of X2 to flow by increasing up-estuary salt flux as freshwater flow increases (Monismith et al. 2002).

Sea level, wind setup along the coast, and atmospheric pressure gradients influence tidal height at the bay mouth and, therefore, tidal currents. Wind-driven circulation over shoals can alter the local salinity gradient, and vertical mixing due to waves set up can break down vertical stratification. These effects are ephemeral and have relatively little effect on the large-scale salinity gradient.

Several trends are likely to alter the salinity gradient. Sea-level rise will move the gradient up-estuary, because deeper water causes increased landward penetration of saline waters on the bottom<sup>1</sup>. This tendency would be enhanced by lower freshwater flow in the dry season and probably opposed by elevated flows during winter. This means that the salinity gradient could move seaward more frequently or further in wet winters than it does now. However, the dry season will be drier and last longer than it does now, resulting in a persistent landward position of the salinity field and persistently high salinity. The net effect on the salinity gradient is very uncertain, mainly because of uncertainties about future freshwater flows.

## Climate

Trends in air temperature with climate change are poorly defined. The air temperature gradient between the ocean and the Central Valley should increase, resulting in a greater pressure gradient driving stronger northwest winds along the coast. These winds in turn may increase the frequency and intensity of upwelling (Snyder et al. 2003), potentially increasing the loading of nutrients and organic matter from the ocean to the bay. Summer wind speed over the Bay might also increase, although model estimates of wind do not support a future increase (Rasmussen et al. 2011). The frequency of large storms has been forecasted to increase together with increasing intensity of the largest (Dettinger 2011).

Water temperature will track air temperature in the upper estuary as it does now (Wagner et al. 2011), but ocean temperature could actually decline (Snyder et al. 2003), so water temperature in the Delta will rise more than in other reaches of the estuary. These effects will result in steepening of the spatial temperature gradient in summer and flattening in winter.

## Nutrients

San Francisco Bay has long been recognized as a nutrient-enriched estuary. However phytoplankton growth and accumulation appear to be largely controlled by a combination of factors, including strong tidal mixing, light limitation due to high turbidity, and grazing pressure by clams (Cloern et al. 2012). These

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<sup>1</sup> i.e. the landward bottom velocity associated with gravitational circulation scales with the cube of water depth for a given slope of the salinity gradient; Monismith et al. 1996.

controls have meant dissolved oxygen concentrations found in the Bay's subtidal habitats are much higher and phytoplankton biomass and productivity are substantially lower than would be expected in an estuary with such high nutrient enrichment (RWQCB 2012).

In the future these controls may be less successful, particularly as turbidity decreases and water temperatures increase. There is a growing body of evidence that suggests the historic resilience of San Francisco Bay to the harmful effects of nutrient enrichment is weakening. Since the late 1990's, regions of the Bay have experienced significant increases in phytoplankton biomass and significant declines in DO concentrations. In addition, an unprecedented autumn phytoplankton bloom in October of 1999, and the 2004 red tide event occurring in the North Bay, further signal changes in the Estuary (RWQCB 2012). Improved understanding of the relative importance of these controlling factors is needed, how they vary around the estuary, how they vary in time and how they may be impacted by climate change.

Another recent issue is the potential impact of ammonia/ammonium on the Bay. Recent studies argue that elevated levels of ammonium limit primary productivity in Suisun Bay (Dugdale et al., 2007, 2012; Parker et al., 2012a), and perhaps elsewhere in the Estuary (Parker et al., 2012b). However, there is currently disagreement within the scientific community about the potential role ammonium plays in limiting primary productivity (RWQCB 2012).

Tidal marshes play a role in improving water quality by cycling nutrients. There is uncertainty as to how bayland restoration will influence nutrient cycling due to the variability and complexity of the system. However, marshes are known to assimilate nitrogen, particularly in the form of nitrate. Wetlands can be highly effective at removing nutrients from wastewater (e.g., Jasper et al., 2014). Marsh restoration helps to transform anthropogenic nutrient inputs to the estuary by retaining and transforming nutrients into less harmful forms. Thus, restoration of marshes may enhance the resiliency of the bay ecosystem with respect to human inputs of nitrogen.

As part of assessing the potential effectiveness and feasibility of this approach, a number of factors need to be evaluated, including:

1. The desired load reduction. Achieving the desired load reduction depends on a number of factors that will in turn have major influences on design considerations, in particular the required wetland area to ensure sufficient residence time and sufficient N removal, and include:
  - a. Whether the same load reduction is required year round or can vary seasonally.
  - b. Seasonal variability in flow and load. Currently, EBDA's flows and loads vary by as much as a factor of 2. Treating high flows would require at least twice as much wetland area as dry flows.
  - c. The form of N (as  $\text{NH}_4^+$  or  $\text{NO}_3^-$ ), and its influent concentration. If N arrives primarily in the form of  $\text{NH}_4^+$ , in general nitrification of  $\text{NH}_4^+$  to  $\text{NO}_3^-$  must occur first, followed by denitrification (unless N loss goes forward by anamox)
  - d. Seasonal variability in removal efficiency due to the effect of temperature. For a given area and flow or loading rate, removal efficiency can vary by more than a factor of 5.

2. Designing wetlands such that they achieve the combined effect of increased wetland habitat, nutrient removal, and wetland accretion to stave off sea-level rise, which may require some compromises on all three fronts to achieve an overall optimum design.

### Human Interventions

Over the time period of the climate forecasts above, several important human interventions are likely to occur or are already occurring. Human population around the estuary will continue to grow, adding impervious surfaces, increasing waste production, and demanding more ecosystem services such as freshwater and recreational opportunities. These demands, particularly for freshwater, could be offset to some degree by increased efficiency of water use, fallowing of lands, or changes in cropping patterns.

The Delta will almost certainly be replumbed either as part of a deliberate, planned process or in response to an emergency such as a massive levee failure (see 2.1E; Lund et al. 2007; BDCP 2013). This could exacerbate the shift in runoff timing, as water will be even more valuable for cities and farms in dry summers.

Despite the increase in human population, nutrient loading from municipal wastewater plants is likely to decrease as the plants are upgraded to meet water-quality standards. This may result in a substantial decrease in ammonium discharge.

Alterations of shorelines will occur both as adaptation to sea-level rise and as part of planned activities to protect and restore wetlands. Restoration activities have begun under the guidance of the Baylands Habitat Goals Report, and the Bay-Delta Conservation Plan also calls for massive wetland restoration in the Delta and in Suisun Marsh (BDCP 2013). This will result in an altered tidal prism and therefore altered tidal heights and currents that will depend on where the alterations occur. Construction of hard structures to protect human uses is likely to increase reflection of waves with unknown consequences for erosion in other locations.

Deliberate or accidental introductions of invasive species will likely continue. Quagga mussels were found in Lake Havasu in 2007 and were established throughout southern California a few years later. Zebra mussels were found in 2008 in a reservoir near Hollister, about 100 km south of the Delta. Both of these highly invasive species are likely to arrive in the freshwater reaches of the estuary despite ongoing efforts to halt their spread. Other introductions are difficult to predict.

There are also a number of events with low annual probability but large consequences. There is a high probability of multiple levee failures in the Delta, resulting in flooding of several highly subsided Delta islands (Mount and Twiss 2005, Lund et al. 2007, Bates and Lund 2013). Such an event, likely driven by an earthquake but also possible through storm-driven flooding, would very quickly increase the tidal prism in the Delta. Reversing this flooding could take years, if it is done at all. If it is not, the result would be a permanent increase in the tidal prism, resulting in greater salinity penetration than occurs now and a shift in currents and spatial distribution of residence times. The actual outcome will depend heavily on how many islands flood, where they are, and when the flooding occurs in relation to the annual hydrologic cycle. The outcome for the lower estuary will be higher salinity at any location and an alteration in tidal currents, which may affect how sediment and organisms are transported around the estuary. Widespread levee failures in Suisun Marsh would have similar consequences. These events have a low probability in any one year but over the time period considered for climate change they may be very likely.

## SUMMARY

The preceding sections describe in some detail the existing drivers of change that are likely to influence future Baylands landscapes. In addition to sea-level rise projections, these drivers also include sediment supply, topography, water quality and supply, nutrients, climate and human interventions.

Sediment supply and its ability to support intertidal Bayland habitats is a critical factor in shaping the future of San Francisco Bay. As sediment supply to the Bay decreases, we will need to carefully match up supply with need to ensure the long term sustainability of our marshes and mudflats.

Human interventions will continue to shape the Bay and its ability to respond to climate change. Region-specific topography and land use will dictate whether we are able to realign levees, restore wetlands and create new upland transition zones, or if we must continue to rely upon traditional engineered flood protection measures.

Regardless of sea-level rise, climate change related shifts in precipitation patterns will influence the amount and timing of freshwater inputs to the Bay. Higher sea levels combined with potentially protracted dry seasons will result in changes to the salinity gradients. These could be exacerbated by levee failures in the Delta.

Changes in other climate variables such as temperature and wind patterns may also affect upland and estuarine habitats and species. Shifts in runoff patterns, nutrient transport, and wind velocities may result in higher water temperatures and changes in turbidity that can increase the potential for eutrophication in the Bay.

Despite these numerous challenges, there are still options available for planners and land managers to adapt to these differences. For example, multi-objective and multi-habitat project designs can maximize cumulative benefits for climate change adaptation. By thoughtfully integrating design features that can have habitat and physical benefits, projects can achieve more than one outcome and increase overall project success by including design components that support each other. The following chapters will examine these strategies to offset the drivers of change.

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# Science Foundation Chapter 2

## Projected Evolution of Baylands Habitats

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## INTRODUCTION

The projected drivers for change, in particular climate and sea level, will likely affect the evolution of baylands habitats over the next century. We understand the baylands to be evolving in three major directions; vertical accretion based on sediment supply and organic accumulation, landward migration (also called transgression) based on availability of terrestrial transition zone space, and lateral movement of the bayward marsh edge. Because tidal marshes are highly sensitive to elevation, their sustainability in San Francisco Bay (and elsewhere) will depend on the balance between sea-level rise and marsh sediment accretion (Michener et al. 1997, Morris et al. 2002). As discussed in Science Foundation Chapter 1, the existing marshes have a range of elevations covering low to high marsh; the higher parts of the marshes give substantial "elevation capital" (Cahoon and Guntenspergen 2010), i.e., they have elevation to lose before they convert to unvegetated mudflat.

This raises two key questions: firstly, how are baylands habitats (e.g., marshes, mudflats and managed ponds) likely to evolve over the next century? To answer this question we need to understand the present evolution and then make projections of future evolution. Once we understand the future evolution we can address the second key question: what management actions can we take to guide the evolution of baylands habitat in the short- and long-term? The decision about when to implement each of these measures will depend on the rate of sea-level rise, and in particular when certain threshold elevations will be crossed that trigger the need for intervention.

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## **HOW ARE THE BAYLANDS HABITATS LIKELY TO EVOLVE OVER THE NEXT CENTURY?**

### Tidal Baylands

There are three ways in which marshes can respond to sea-level rise affecting the area and distribution of habitat, and potentially leading to the transformation of the habitat type:

- 1) accrete vertically, with no net loss of habitat if they keep up with sea-level rise;
- 2) migrate upslope as sea levels rise and convert terrestrial areas to wetlands; and,
- 3) if wetlands cannot keep pace with rates of sea-level rise, convert or “downshift” from mid to low marsh, or from marsh to mudflat and subtidal.

Key to how a marsh will evolve with rising sea level will be how much “elevation capital” the marsh has. This determines the inundation regime and the accretion rates that maintain the “elevation capital” and so determine the evolution of the inundation regime. As discussed in Science Foundation Chapter 1, theoretical feedback models predict that changes in salt marsh plant production and elevation will be observed in response to the position in the tidal frame and hence the inundation regime (Morris et al. 2002). This model suggests an inundation regime that maximizes sediment accretion and productivity and also represents the upper limit of flooding tolerance for a given plant species. Maintaining that elevation capital with accelerating rates sea-level rise requires increased accretion rates. Low accretion rate leads to loss of elevation capital which may lead to changes in the inundation regime, anaerobic stress levels throughout the wetland, and shifts in habitat types (Rybczyk et al 2013).

### Accretion Rates

Fundamental to the resiliency of marshes and maintenance of elevation capital is their accretion rate. Presently, tidal wetlands within the Bay area are accumulating enough sediment to keep pace with sea-level rise. For example, observations by Callaway et al. (2012) who measured historic accretion rates at six tidal marshes in San Francisco Bay (including South Bay, North Bay and Suisun) show in all cases, accretion rates from mid- and high-marsh station were close to 3-5 mm/yr, with little difference in accretion rates across sites. At many sites, accretion rates were slightly greater in the low marsh locations (Callaway et al. 2012), and these patterns of greater accretion at sites closer to the Bay mirror patterns found in many other tidal marshes across the world (e.g., Hatton et al. 1983, French et al. 1995). However accretion at individual sites varies according to local conditions. Other local variations have been reported by Drexler et al. (2009) at Browns Island, where lower rates were observed due to compaction. In Coyote Creek, Watson (2004) found much higher rates associated with rapid marsh expansion along the creek. Patrick and DeLaune (1990) also documented much higher accretion rates at the adjacent Alviso Slough. The unusually rapid accretion in this area was likely influenced by very high local, short-term subsidence rates associated with earlier groundwater withdrawal for agriculture in the South Bay couple with the abundant fine sediment supply in the far South Bay (Patrick and DeLaune 1990).

Rates of sediment accretion at newly restored sites are often substantially greater, especially at sites that are subsided, as mineral sediment inputs are much greater at lower elevations (Williams and Orr 2002, PWA and Faber 2004). For example, Callaway (unpublished data) has measured rates greater than 10 cm/yr at the Island Ponds (Ponds A19, A20 and A21, breached in 2006 as part of the South Bay Salt Pond Restoration project), and even more rapid rates at Pond A6 (breached in 2010), which started at lower

elevations than the Island Ponds. Accretion rates from restoration sites that are at elevations similar to natural marshes (e.g., the upper zone of Muzzi Marsh) are similar to those found in natural tidal marshes in the Bay, while accretion rates for restoration sites at lower elevations are intermediate between the high rates from highly subsided sites and those from natural marshes (Callaway, unpublished data from sites along an elevation gradient at Muzzi Marsh). As restoration sites build to elevations more typical of well-developed natural marshes, accretion rates slow down over time (Williams and Orr 2002).

Projecting accretion rates into the future requires the modeling of the complete accretion processes as described in Science Foundation Chapter 1 together with projections of sea-level rise and sediment availability. A number of models have recently been used to assess the evolution of tidal marshes in the San Francisco Bay. These include MARSH98 (Stralberg et al., 2011), WARMER (Takekawa et al., 2012; Swanson et al., 2013), MEM (Schile, 2012), and an empirical model of accretion (Thorne, 2012) (see Appendix 2.1 for details). Each model has been developed to address different aspects of marsh accretion and have been applied in different locations around the Bay. All the models record the elevation of a point within the intertidal zone from the accretion of sediment and organic matter dependent on suspended sediment availability and plant productivity (Figure 2.1). In the cases of WARMER and MEM, the compaction and decay of the soil are also explicitly included.

All of the results from these models are sensitive to the rate and magnitude of sea-level rise and the supply of sediment to the marsh. Both of these variables have uncertainty in their future values. The models were found to be less sensitive to the different organic accumulation scenarios tested.

The MARSH98 modelling (Stralberg et al., 2011) has the most comprehensive geographical coverage of the Bay. The study looked at existing marshes and also areas with potential for landward migration or restoration following dike removal. Across all sea-level rise and sediment supply scenarios, the models project an increase in mid-marsh habitat between 2010 and 2030 throughout the estuary. In each of the scenarios this increase comes at the expense of high marsh and terrestrial habitat. (Figure 2.2).

Between 2030 and 2050, the model projects that the area of low marsh habitat will increase and the area of high marsh and upland will decrease across all scenarios; for the high sea-level rise/low sediment scenario the area of mid marsh habitats throughout the estuary will also decline. In general, the area of tidal marsh habitat was projected to remain relatively unchanged between 2030 and 2050, but the composition of the marsh habitat changed, with the amount of low marsh habitat projected to increase and the amount of mid marsh habitat projected to decrease.

The model projects an increase in the total estuary wide area of mid-marsh habitat from 2010 for low sea-level rise scenarios for either sediment assumption and for the high sea-level rise/ high sediment assumption. However, the model also indicates that more than 90% of mid-marsh and high-marsh habitat will be converted to low-marsh, mudflats or subtidal habitats by 2100 in the high sea-level rise/low sediment scenario. The models do indicate that there are opportunities for unimpeded landward migration of marshes, with 5,000-7,500 acres of currently terrestrial habitat potentially evolving to tidal marsh by 2100 depending on the scenario.

The potential impact on specific marshes can be seen in the model projections of China Camp undertaken by Swanson (2013) and Schille (2012). Increasing the rate of sea-level rise and decreasing the availability of sediment increases the potential of habitats downshifting (Figure 2.3).

The NRC (2012) report shows a range of 40 cm at 2050 for the amount of sea-level rise and about 1.2 m at 2110. This range has huge impacts for what happens to the marshes. The sensitivity analysis of the MARSH98 and WARMER models (Stralberg et al. 2011, Swanson et al. 2013) indicated that these two parameters largely control the final elevation of the marsh for the range of conditions considered in those simulations. The model simulations agree that with low rates of sea-level rise (e.g., 52 cm/century), the marsh elevations can keep pace with sea-level, even with low sediment availability. However, with sea-level rise rates greater than 100 cm/century and low sediment supply there will be a decline in mid and high marsh habitat.

### Downshifting of Habitats

If sea-level rise continues to accelerate, at some point it may outpace the rate of accretion and the marsh will start to ‘drown’. If the vertical accretion of marshes cannot keep pace with sea-level rise then the wetlands habitats will tend to migrate landward. The horizontal rate of migration will depend upon the rate of sea-level rise and the slope of the transition zone. Historic diking steepened coastal gradients around the Bay, converting gently sloping bayland edges that rise towards the land into steep linear borders backed by basins. Sea-level rise acts very differently depending on the gradient of its landward boundary. On gentle, continuous slopes it gradually shifts tidal habitat zones landward and landward, while on discontinuous, artificial diked bayland topography it forces either acceleration of maintenance and repair of dikes, or “overstepping” the barrier – abruptly flooding the diked basin and radically shifting the shoreline and shore processes landward. If the marsh is bounded by a steep slope (such as an inboard levee) then the transition zone available for migration will be much reduced and marsh habitat will be lost through ‘coastal squeeze’.

There are a number of (qualitative) evolutionary scenarios for the baylands:

- a. **Equilibration, dynamic stability:** existing tidal marshes accommodate sea-level rise with only minor long term or progressive conversion of tidal habitat types, and a gradual landward shift (horizontal displacement or landward estuarine “migration”) in position. This scenario is associated with very gradual (historic) rates of sea-level rise and net positive sediment budgets (due in part to effects of diking, artificial loss of tidal prism). This scenario is not likely to occur in a regime of rapidly accelerating sea-level rise and neutral or negative sediment budgets.
- b. **Gradual evolution:** gradual submergence of tidal marsh habitats with marsh type conversion (“downshifting” zones: high marsh to middle marsh, middle to low, low marsh to unvegetated tidal flat); expansion of tidal marsh pans and enlargement of tidal channels; mudflat erosion (loss of elevation); progressive but slow erosional retreat of marsh edges (wave-cut marsh “cliffs” or scarps); and either dike overtopping, erosion, and breaching, or dike raising, armoring, and increased artificial bayland drainage. The “gradual evolution” bayland scenario is compatible with coastal planning adaptation through modification of Baylands.
- c. **Collapse** (abrupt conversion of ecosystem to alternative modes and habitat types): in this worst-case scenario associated with early onset of accelerated sea-level rise at the upper end of projected rates, sea-level rise would overstep marsh platforms, causing wholesale drowning of marshes: marsh plains initially respond by converting to low marsh (cordgrass), but founder as rapid marsh vegetation dieback forms extensive pans that “swallow” fragmented marshes and expand to tidal flats. This is analogous with contemporary tidal marsh loss in Elkhorn Slough, Gulf of Mexico and the Mississippi Delta. Rapid marsh edge and levee erosion, increased flooding of diked baylands or undiked adjacent lowlands, and the rapid loss of critical high marsh habitat and terrestrial buffer integrity are likely to occur in this scenario.

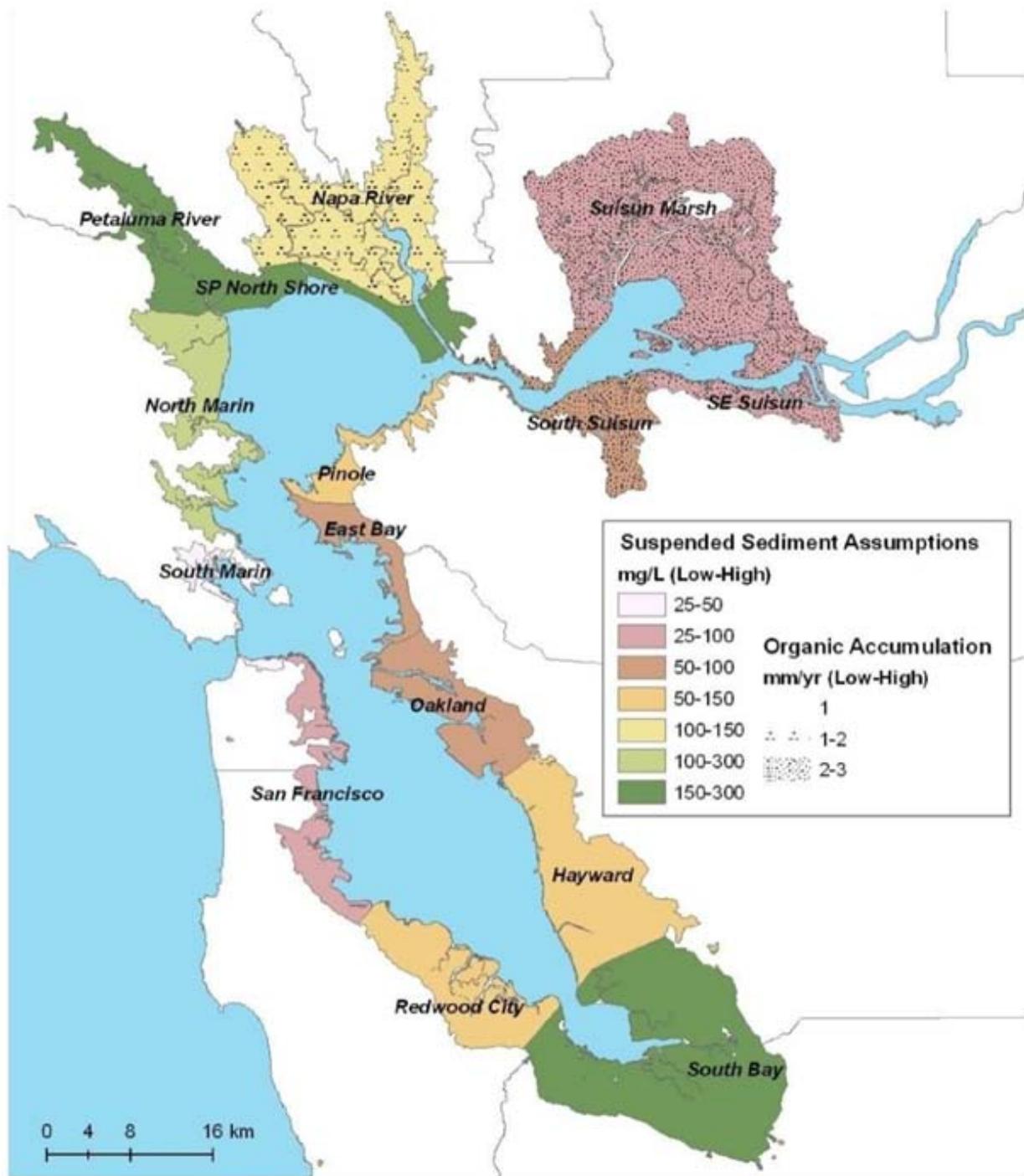
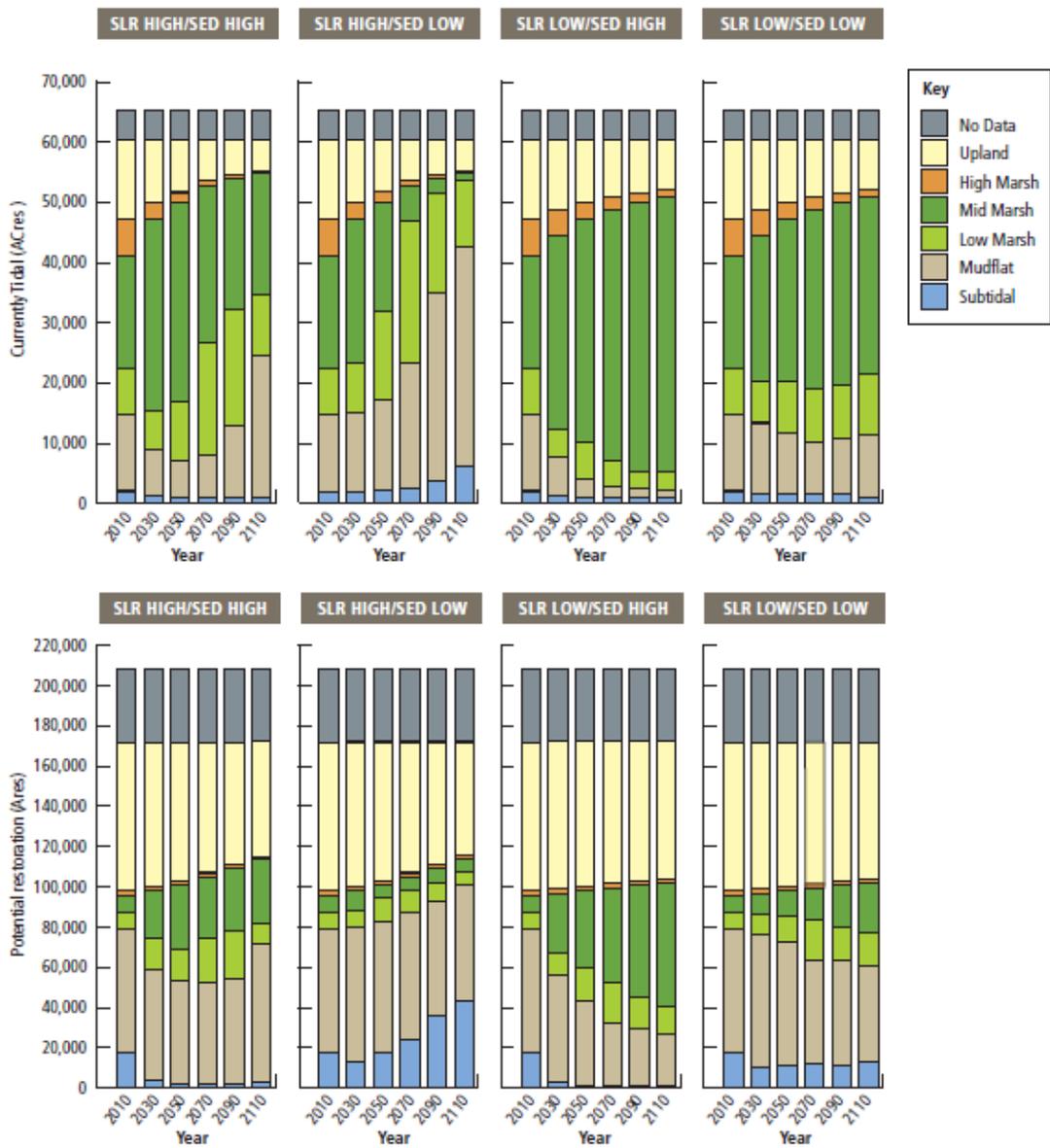
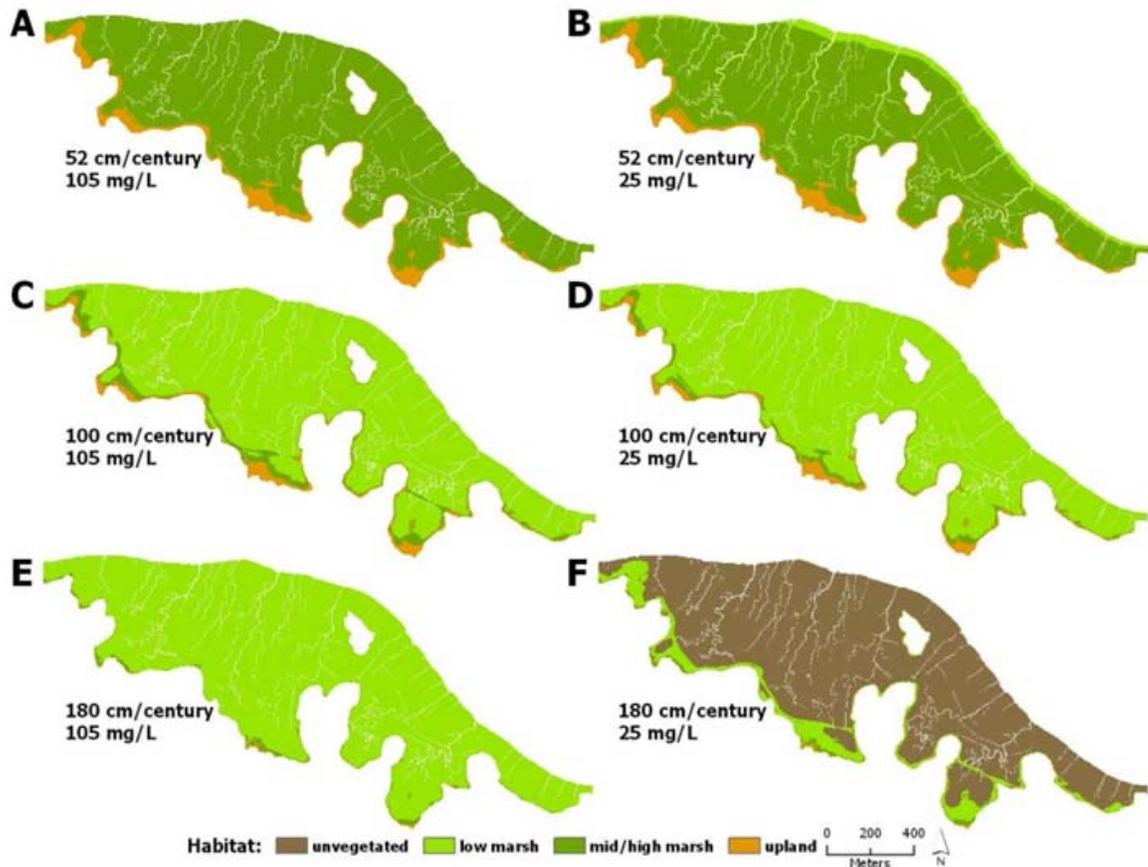


Figure 2.1. Biogeomorphic sub-regions within San Francisco Bay study area and assumptions about suspended sediment concentrations and organic matter accretion rates for climate change scenarios (Stralberg et al., 2011).



**Figure 2.2.** Results from the Marsh98 model showing projected marsh habitat extents under different sea-level rise (SLR) and sediment supply (SED) scenarios for both current tidal areas and potential restoration areas. Note the different y-axis scales. Adapted from Stralberg et al. (2011).



**Figure 2.3.** MEM scenarios results at China Camp Marsh for various rates of sea-level rise and available suspended sediment concentrations (Schille 2012).

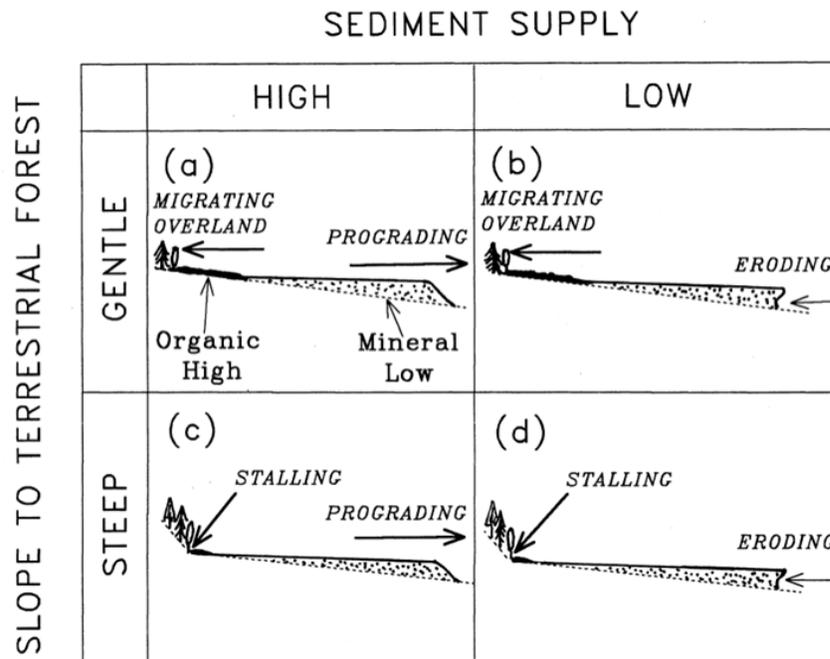
There will probably be a variable mix of scenarios a) and b) for the first 50 years, unless there is an abrupt, rapid acceleration in sea-level rise (i.e. abrupt changes in ocean temperature or ice sheet collapse). Maintaining existing marsh zones with no conversion would be an optimistic projection because as marsh plain drainage decreases with submergence, so does marsh plant growth and vegetation height. Reduced marsh vegetation growth will mean less stem height and density for trapping and stabilizing suspended sediment and less production of organic matter in the soil profile.

The major controlling variables are rate of sea-level rise, sediment supply and space for migration, only two of which we can influence. Increasing the sediment supply by supplementing sediment onto, or close to, the wetlands may increase the local accretion rate. Increasing the terrestrial buffer area and reducing the inboard levee slope will increase the space for migration.

Brinson et al (1995) provides a summary framework for addressing the transformation from one habitat class to another as sea-level rises; from uplands through wetlands to mudflat and subtidal. This is illustrated in Figure 2.4 where estuarine-terrestrial interaction can be described as one of four possible interactions based on two conditions at the transition zone edge (migration versus squeeze) and two at the marsh edge (prograding versus eroding) for both low and high sediment availability (which generally equates to high and low accretion rates in the Bay).

This framework simplifies terrestrial-estuarine interactions into four combinations based on sediment supply and vertical accretion rate. There two possible conditions at the terrestrial-marsh margin (i.e., overland migrating versus stalling) and two at the marsh-estuarine margin (i.e., prograding versus eroding). These four patterns shown in Figure 2.4 are:

- 1) **migrating** overland and **prograding** toward the estuary - sediment surpluses convert subtidal estuary into low marsh. As sea-level rises, mineral low marsh and organic high marsh encroach on adjacent terrestrial areas by migration; tidal creeks and deltas are prograding toward the estuary because sediment is abundant.
- 2) **migrating** overland and **eroding** away from the estuary. Rapid submergence due to a combination of sediment starvation and accelerated rates of relative sea-level rise. Loss of marsh to subtidal estuary is both a marsh edge phenomenon and the drowning of the marsh surface.
- 3) **squeezing** at the terrestrial margin and eroding toward the estuary, and
- 4) **squeezing** at the terrestrial margin and **prograding** toward the estuary. (c) and (d) are extremes of a continuum where overland migration has been virtually halted by local contact of the marsh with a steep slope.



**Figure 2.4.** Classes of the response of marshes to sea-level rise as a function of extremes in sediment supply and landward slope, after Brinson et al (1995).

Whether the terrestrial margin of marshes is migrating or being squeezed is largely a function of the vertical accretion rate and the steepness of the slope in the transitional area at this margin. Net change in surface area in a squeezed marsh is the balance between vertical accretion, erosional losses and seaward gains.

Combining the possible combination of conditions gives a description of how the marsh may evolve. For example, a low sediment supply and shallow slope lead to marsh edge erosion and headward incision of tidal creeks into marshes, and are accompanied by migration of the marsh-terrestrial interface.

The pace of coastal habitat changes due to sea-level rise, even in “gradual” scenarios, may not be uniformly gradual. Average sea level represented in models deviates significantly from annual fluctuations in sea level, which may reach up to approximately 8 inches above average levels during strong El Niño events due to thermal expansion of warm Pacific waters. In addition, intense storms also associated with El Niño events also may be expected to achieve many years or even decades’ worth of “average” erosion in extreme storms or series of storms. Thus, the coastal habitat changes expected with sea-level rise, regardless of the long-term sea level curve, may not be expected to occur in a linear or incremental pattern. The biological responses to habitat change caused by sea-level rise may similarly be expected to occur in pulses, or reflect dominant influences of extreme storm events. Local extirpation of species with limited dispersal ability, high tide refugia fidelity, or close dependence on narrowly distributed critical habitats, is a particular concern for threshold changes in habitat driven by storm events during long-term sea-level rise.

### Diked Baylands

Many parts of the Bay are constrained in some way and their ability to evolve as described above is limited. Diked baylands exist in parts of the Bay that once were tidal but are now isolated from the tides and sediments. “Reclamation” typically involved the construction of earthen berms along the margins of the marsh plains where they bordered mudflats or large tidal channels. In the original Baylands Goals report, key diked bayland habitats described included diked wetland, agricultural bayland, salt pond, and storage/treatment pond. These types of diked baylands vary in their distribution around the Bay, with salt ponds largely located in the North and South Bay, waterfowl ponds in Suisun Bay, agricultural baylands in the North Bay, and water treatment ponds in the Central and South Bay.

Although diked baylands are not natural features of the Bay, some of them do provide significant habitat values (e.g., diked wetlands, salt ponds). In this update, we will refer to managed ponds as those diked baylands that are physically separated from the tides by a berm or levee, and have artificially controlled water levels and/or salinities through a water control structure (e.g., weir, culvert, flapgate, etc.) for the purposes of enhancing wildlife values. Despite the original (non-ecological) intent of some of these diked baylands such as salt ponds, ancillary wildlife benefits have been observed. More recently, attempts are being made to enhance or maximize the wildlife value of these managed ponds. The management of water levels and salinities will vary depending on the target species as well as the pond’s location within the Bay. For example, management of a pond for diving ducks may include deeper water at lower salinities, while small shorebirds would require shallower water for roosting and foraging. Certain ponds may also require seasonal shifts in management to maximize benefits to target species. In addition, certain parts of the Bay are more deeply subsided, and therefore the infrastructure required to manage a specific pond will vary depending on its average bed elevation relative to the tides.

In addition to managed ponds, the Bay also supports numerous diked baylands that are managed as emergent vegetated marshes. These include diked and muted tidal marshes, treatment wetlands, and mitigation wetlands. The target habitat and wildlife species vary greatly, from large duck clubs, to small mitigation projects to support endangered species such as the salt marsh harvest mouse. These managed marshes are found throughout the entire Bay, and were designed to optimize the habitat within their footprint, with sometimes little consideration to their place in the landscape. The resulting habitats range from large, connected systems like in Suisun Bay, to small disparate marshes that are isolated from similar

natural habitats. Many of these marshes also carry with them specific regulatory constraints, such as discharge standards, or minimum acreage requirements.

In addition to managed ponds and managed marshes, there are significant areas of other types of diked former baylands. Most significant among these are urbanized areas. Large areas of the Bay's edge was historically diked and converted to urban and suburban uses, including airports and entire cities. In addition, significant areas were similarly diked off and converted to agricultural uses. Agricultural uses vary greatly by subregion, and range in value from low (hay fields) to high (vineyards). Other, smaller diked bayland types are also present and include areas such as water treatment ponds, playa areas, and even vernal pool complexes.

When evaluated in the context of climate change, managed bayland habitats face a somewhat unique set of considerations for their long-term viability. Similar to the maintenance required of engineered levees for flood control features, the impact of sea-level rise on infrastructure associated with managed habitats will be similarly impacted.

It is likely that restored and resilient tidal marshes may not provide all of the habitat functions currently provided by managed ponds. It would however be prudent to minimize the reliance on managed systems to the maximum extent possible given the ecological goals of the Bay, as these ponds become spatially fixed features in a Bay that is dynamic and moving landward. They will become increasingly more difficult to maintain and operate as the bayland habitats around them migrate and change, and therefore should be located in areas that facilitate long-term viability as well as ease of operations and maintenance. Target wildlife species (usually birds, as fully tidal systems are generally better for fish and mammals) typically use these managed ponds for roosting, foraging and nesting. Trying to maximize these ecosystem services in the landscape requires not only considering the best location for managed ponds, but also their proximity to other resources (such as tidal mudflats for foraging) needed by these species. However, innovative ideas to make managed ponds more resilient (e.g., flexible water management, sediment capture inside the ponds, etc.) should be included in any new managed pond investments.

To adequately control water levels and salinities inside the ponds for a target suite of species, engineering considerations for the water control structures and surrounding levee systems usually require specific elevations for water intake and outlet points. Source waters for managed baylands may be the bay itself, or adjacent freshwater sources, depending on the location and habitat goals of that pond. Climate change related stressors such as higher water surface elevations, increased frequency and intensity of storm events, and regional salinity shifts will all impact the ability of managers to maintain target habitat conditions inside the ponds. For example, higher salinities in parts of the bay may mean that target salinities inside the managed pond may become harder (or easier, depending on the target suite of species) to maintain.

As water levels and salinities change, several possible management actions may be needed to sustain target habitat conditions inside the ponds. In the long term, the integrity of levees will come under pressure either due to increased overtopping of the crest, or direct erosion of the levee itself. The most immediate action would be to raise or reinforce existing levees to keep unregulated tidal waters out and retain the ability to control internal water levels (Figure 2.5). Therefore, similar to the relationship between levees and marshes, managed baylands should be located in positions on the landscape that take advantage of the protections afforded by outboard tidal marshes, or other site specific considerations.

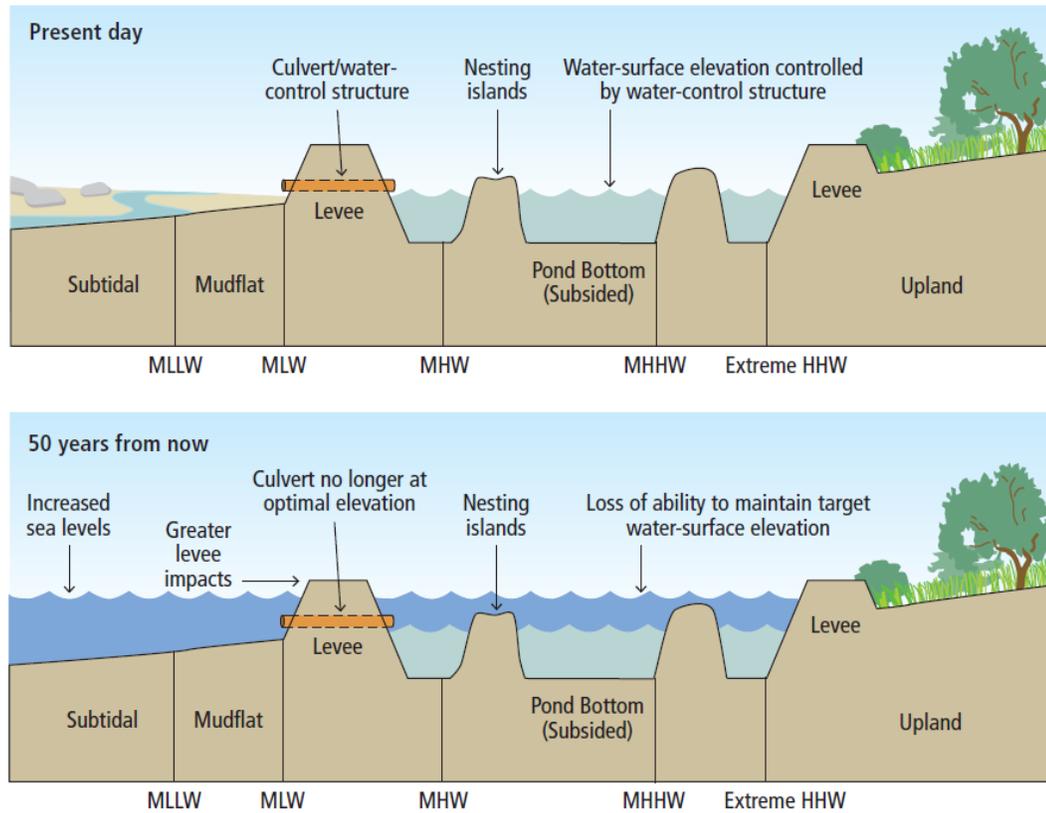


Figure 2.5. The impact of sea-level rise on managed ponds over time.

Furthermore, to sustain water management capabilities, it may require changing the management regime of the water control structures. However, it is more likely that water control structures would have to be modified, added, or replaced, and managed ponds may become more reliant on pumping of water as opposed to more passive gravity-driven configurations. In more extreme cases, a managed retreat scenario may be appropriate for some of these ponds, requiring relocation or abandonment of ponds in areas of higher threat from sea-level rise. Abandoned ponds could then be converted to other (likely tidal) habitat types dependent upon their elevation and location in the Bay. In addition, there should be large scale and long term planning between the regions of the Bay and Delta to ensure that the habitat needs of a variety of species are being met regionally as habitat types shift from restoration actions, salinity changes, etc.

## WHAT MANAGEMENT ACTIONS CAN WE TAKE?

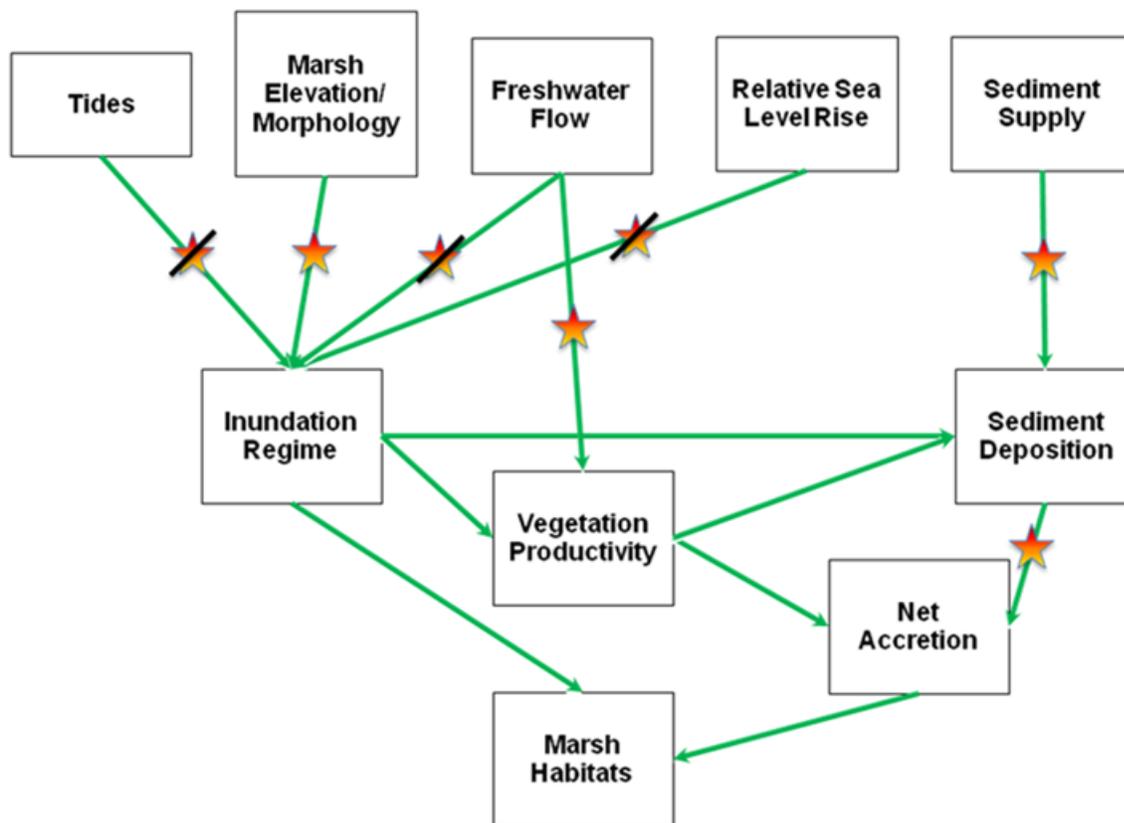
Given the projections of marsh loss in the latter parts of the century under the likely scenario that sea-level rises rapidly and sediment availability is low, we need to understand the range of management actions we can take to guide the evolution of baylands habitats in the short term and in the long term. The major drivers of change of tidal marsh evolution are the rate of sea-level rise (i.e., water depth), inorganic sediment supply, organic productivity, incident wave energy, and space for migration. These in turn control the following key factors of marsh evolution:

- rate of vertical accretion,
- rate of horizontal erosion; and
- rate of landward migration.

### Vertical Accretion Measures

The purpose of the set of measures described below is to increase the vertical accretion rate of the marsh by increasing the supply of fine sediment to the marsh, by improving the pathways by which the sediment crosses the marsh or by increasing the trapping of sediment on the marsh. The intent is to accelerate inorganic accretion rates to keep pace with rising sea level, allowing the marsh to maintain its vertical position in the tidal frame.

A geomorphic process model for vertical accretion in tidal marshes is shown in Figure 2.6. The rate at which marshes accrete vertically is in part a function of the inorganic sediment supply, which in turn is a function of the average suspended sediment concentration in the water column, depth of water and period of high water and in part a function of organic productivity. For inorganic sedimentation, the higher the concentration and deeper the water over the marsh, the greater the amount of sediment available in the water column to be deposited. Marsh accretion rates can be directly affected by increasing the supply of fine sediment in the water column, either by introducing fine sediment directly into the water column (water column recharge) or by placing sediment on the mudflat that may be later resuspended by wave action (mudflat and marsh recharge) to be carried on to the marsh and deposited by natural processes. Sediment supply can also be enhanced by ensuring that there are sufficient size and density of tidal channels to convey fine suspended sediment from the Bay to the landward portions of the marshes. Significant accretion can occur during extreme events when water levels are high and there is significant wave activity to resuspend the bay sediments. Trapping efficiency of fine sediment can be improved by increasing the density of vegetation by planting or by emulating vegetation with the construction of sedimentation fences or similar features. These measures are summarized in Table 2.1.



**Figure 2.6.** Conceptual model of vertical accretion. The stars indicate linkages that could be influenced by management measures. Stars with a bar across them are linkages where management measures are impractical.

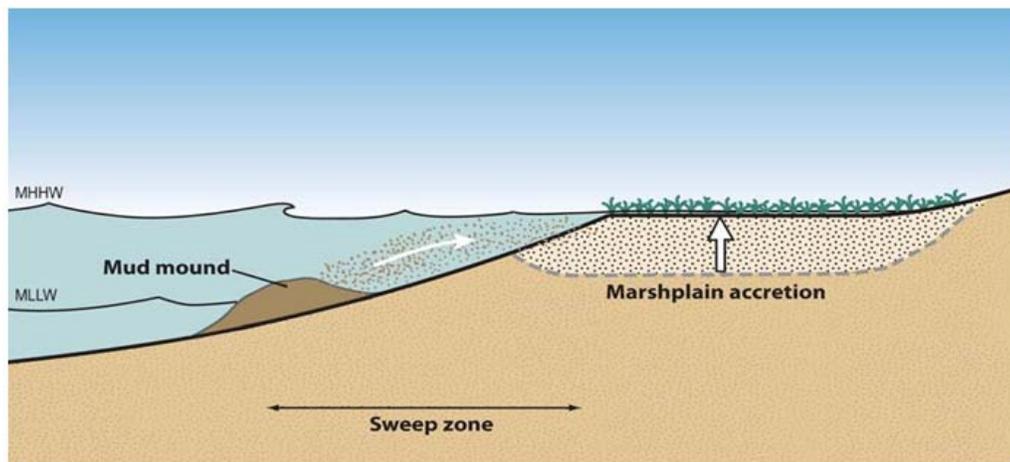
**Table 2.1.** Vertical accretion management measures

Key Factor	Driver of Change	Measure	Examples
Vertical Accretion	Sediment Supply from fluvial and oceanic sources	Mudflat and marsh recharge	Mudflat and marsh charge
		Improve sediment pathways	Improve channel network
	Sediment Deposition	Enhance sediment trapping	Sedimentation fences

*Mudflat and Marsh Recharge*

A number of methods have been suggested to increase the local concentration of fine sediment in the water column. As an example, typical suspended sediment concentrations in the bay are of the order of 100-200 ppm and typical accretion rates of 2-5 mm per year (Callaway *et al* 2012) while observations of accretion rates in Alviso Slough in the South Bay have shown that concentrations of 450-600ppm result in inorganic accretion rates of up to 1-2 feet per year (Ruth and Going 1980 in PWA 2005) while brackish marsh vegetation that supports wildlife is maintained on the marsh surface. By focusing the introduction of fine sediment close to the target tidal marsh, it should be possible to increase the local marsh and mudflat accretion rates. The intent is to make use of natural wave action and tidal currents and watershed outflow to transport the sediment into and deposit it onto the marsh. This avoids both the difficulty of mechanically placing sediment to accurately mimic a natural marsh and the impact of construction equipment on the existing (or restoring) marsh. However there are detrimental impacts from some of these approaches (e.g., smothering benthos on the mudflats) that need to be evaluated.

To introduce sediment directly into the water column, fine sediment could be pumped from a barge adjacent to the tidal marsh. The release would occur on a flooding tide that would carry the sediment to the marsh. An alternative method can be used where the site is exposed to moderate wave action. Fine sediment is introduced indirectly into the water column by placing the material on an adjacent mudflat as a mound either by split bottom barge, hydraulic pipe or ‘rainbowing’. Wave action on the mudflat mound then re-suspends the fine sediment into the water column and conveys it into the tidal marsh (Figure 2.7). In addition, sediment sources from the landward side of the Bay should also be considered for sustaining marshes, either from direct placement of clean fill material into subsided areas to be restored, or enhancing natural sediment pathways through watershed level enhancements and the reconnection of creeks and rivers to the marsh.



**Figure 2.7.** Recharge of marshes.

### Opportunities and Constraints

Recharging the water column and the mudflats has considerable benefits as it allows the choice of when, where and how much sediment to introduce to the system in a targeted fashion to offset some of the overall reduction in suspended sediment forecast for the next century (Schoellhamer 2011). Many of the ecosystem services of a tidal marsh are a function of elevation and inundation regime and so are dependent upon the marsh maintaining its position in the tidal frame. Recharge will help maintain tidal marsh ecosystem services (such as wave attenuation) longer with rising sea levels by increasing vertical accretion of the mudflat and tidal marshes.

However, mudflat, water column recharge and reconnecting creeks to marshes are untried in the Bay and present significant permitting challenges as they could have detrimental impacts particularly on existing habitat. Once the recharge has occurred there is little control on where and how much sediment will be finally deposited as it is controlled by natural processes; the location and timing of recharge is therefore very important. Mudflat recharging with fine sediment will also result in the burial of existing mudflat habitat and associated impacts on species using that habitat. Finally, recharge is likely to work best with relatively small volumes released at frequent intervals – the recharge would be scaled to the marsh area and long term deposition rate. This may conflict with the availability of dredged material, which may occur more infrequently and in larger volumes. Pilot projects to determine the efficacy of this approach should be conducted and studied.

### *Improve Sediment Pathways*

Tidal channels are important elements in tidal marshes as they act both as pathways for sediment, organisms, detritus etc and also as habitat for fish and birds. Mature natural marshes tend to have extensive and complex dendritic drainage networks with several orders of sinuous channel, often divided into watersheds with tidal slough connections to the Bay. These channels convey turbid Bay water into the marshes and on to the marsh plain at high water allowing sediment deposition to occur across the whole marsh. Distance from channel is important in determining the rate of accretion. Coarse material is deposited closer to the channels and forms natural levees which are higher than the surrounding marsh plain and support particular plant species such as gumplant (*Grindelia*). Finer sediment is deposited further away from the channel, diminishing with distance away from the channel and organic material becoming more important. If the channel density is low, parts of the marsh may be far from channels, have few natural levees, be poorly supplied with fine sediment and have low rates of accretion.

The characteristics of channels and channel network are determined in large part by the tidal prism of the marsh, which in turn is controlled by the tidal range and the area and elevation of the tidal marsh. If there is insufficient tidal prism, due to filling of the marsh or diking, then channel networks may not fully evolve resulting in poor habitat and low accretion rates at the back of the marsh away from the channels (Figure 2.8). The relationship between tidal prism and channels can be predicted using hydraulic regime theory (PWA 2002, Williams et al 2002).

### Opportunities and Constraints

Creating a complex channel system that is configured for the size of the marsh will improve the connectivity of the marsh with the estuarine tidal processes, allowing tidal access to all parts of the marsh. This will increase the tidal prism which in turn will allow larger channels to be maintained as well as increasing the volume of sediment available for deposition on the marsh. Many species of plants and animals also rely on channel bank habitat. Increasing the sinuosity of channels provides more

heterogeneity in the habitat. A complex drainage system with a variety of channel orders provides a variety of channel sizes, elevations and inundation regimes for different species.

When tidal action is reintroduced to a subsided site, tidal flows tend to concentrate in existing ditches or depressions that then fix the location and shape of the tidal drainage system. It is likely that once the existing drainage system captures the tidal flows this pattern will persist. Modifications are often made to the drainage, such as the creation of ditch blocks, to avoid such capture.



**Figure 2.8.** Muzzi Marsh showing difference in channel density with elevation of the site due to filling.

### *Enhance Sediment Trapping*

Increasing the period of sedimentation during a tidal cycle may be achieved by decreasing transport velocities. The rate of deposition is controlled largely by the interaction between tidal current velocities and vegetation cover. One example is traditional fine-grained sediment-deposition (‘warping’) methods where high tide waters are impounded and sediments allowed to settle, before draining the waters off (‘dewatering’) via a sluice gate, or as the tide falls. Another method is the use of sedimentation fences which are brushwood structures designed to slow the passage of water thereby enhancing the deposition of sediment. Brushwood fences can be arranged on the shore either as groins or polders.

### Opportunities and Constraints

This measure is largely focused on managed ponds that have yet to be restored, and takes advantage of natural tidal flow to trap fine-grained alluvial sediments across areas of mudflat and saltmarsh. Sedimentation fences either constructed as groins or polders require intensive management and maintenance and so these techniques may be only feasible for relatively small marsh areas and must take local sedimentary trends into account to be successful. This could be combined with shoreline stabilization methods (described below) that reduce waves and capturing resuspended sediment with plantings behind the wave breaks.

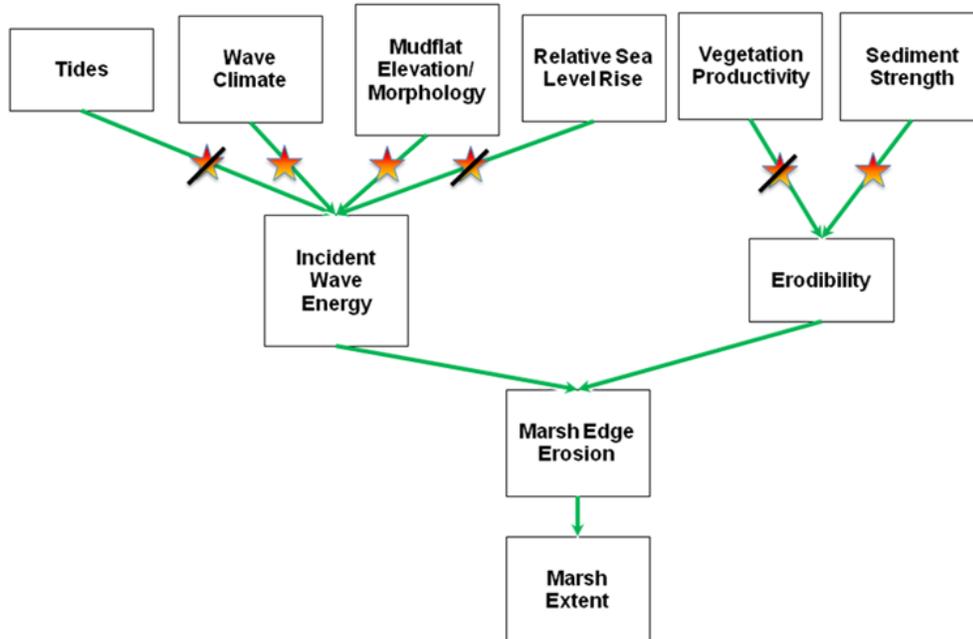
### Shoreline Stabilization Measures

The purpose of the measure is to decrease the horizontal erosion rate of the marsh by reducing the incident wave energy or increasing the stability of the shoreline. The intent is to slow the loss of tidal marsh due to erosion, allowing the marsh to maintain its width for longer.

A geomorphic process model for horizontal erosion in tidal marshes is shown in Figure 2.9. Marsh erosion rates can be directly affected by decreasing wave energy on the marsh edge. This can be achieved by increasing the wave attenuation over the mudflat by increasing its elevation, by increasing the bottom friction of the mudflat by planting submerged aquatic vegetation and by constructing low-crested breakwaters or berms/sills, including living shoreline elements such as shellfish reefs. The stability of the marsh edge may be increased by armoring it with a beach constructed of relatively coarse material and stabilizing the beach with control structures such as groins and headlands constructed with large woody debris (LWD) or rock. These measures are summarized in Table 2.2.

#### Coarse Beach Stabilization

Coarse gravel beaches are a natural form of shoreline that can adjust to local wind-wave conditions and water levels even under conditions of extreme wave events. Unlike typical engineered revetment systems, such as rip rap, movement of cobble and gravel are an inherent characteristic of a coarse beach and not an indication of failure. Dunes fronted by composite gravel beaches experienced erosion rates that were typically 20-40% of pure sand beaches (Allan et al 2005), highlighting the level of protection offered by a gravel beach as compared with a pure sand beach. In Southern California, researchers have noted that gravel beaches tend to gain material and increase their crest elevations during severe storms, while adjacent sand beaches eroded (Lorang et al. 1999, Everts et al. 2002).



**Figure 2.9.** Conceptual model of horizontal erosion. The stars indicate linkages that could be influenced by management measures. Stars with a bar across them are linkages where management measures are impractical.

**Table 2.2.** Horizontal erosion management measures.

Key Factor	Driver of Change	Measure	Examples
Horizontal Erosion	Erodibility	Coarse beach stabilization	<ul style="list-style-type: none"> <li>• Coarse sediment pocket beaches</li> <li>• Crenulate shoreline - rock/LWD headland</li> <li>• Sand/shell barrier beach</li> </ul>
	Wave Climate	Wave attenuation	<ul style="list-style-type: none"> <li>• Static rock berm</li> <li>• Dynamic gravel/cobble berm</li> <li>• Oyster reef/eel grass beds</li> </ul>

### Opportunities and Constraints

Marsh erosion due to wave action can cause narrowing of marshes and compression of wildlife habitats closer to terrestrial edge buffers. Gravel beaches are one of the most effective forms of coastal protection, exhibiting significant stability under sustained wave attack (Ahrens 1990, Ward and Ahrens, 1991). The sloping, porous coarse beach, once prevalent in the Central Bay, is able to dissipate the wave energy by adjusting its shape in response to the prevailing wave conditions. This approach would provide the geomorphic foundation for gradual migration and ecological transition of native vegetation and habitats associated with the shoreline.

A major constraint that likely limits the adoption of coarse beaches as a viable option is the identification of suitable gravel sources that could be utilized in the construction and maintenance of such structures. Few natural sources are available in the Bay. In addition, the potential impact of longshore drift should be considered in any beach design and low groins, constructed across the gravel beach, may be necessary to reduce the rate of longshore gravel transport. The placement of gravel and cobble on the shoreline would also lead to the conversion of habitat types and effects on benthos.

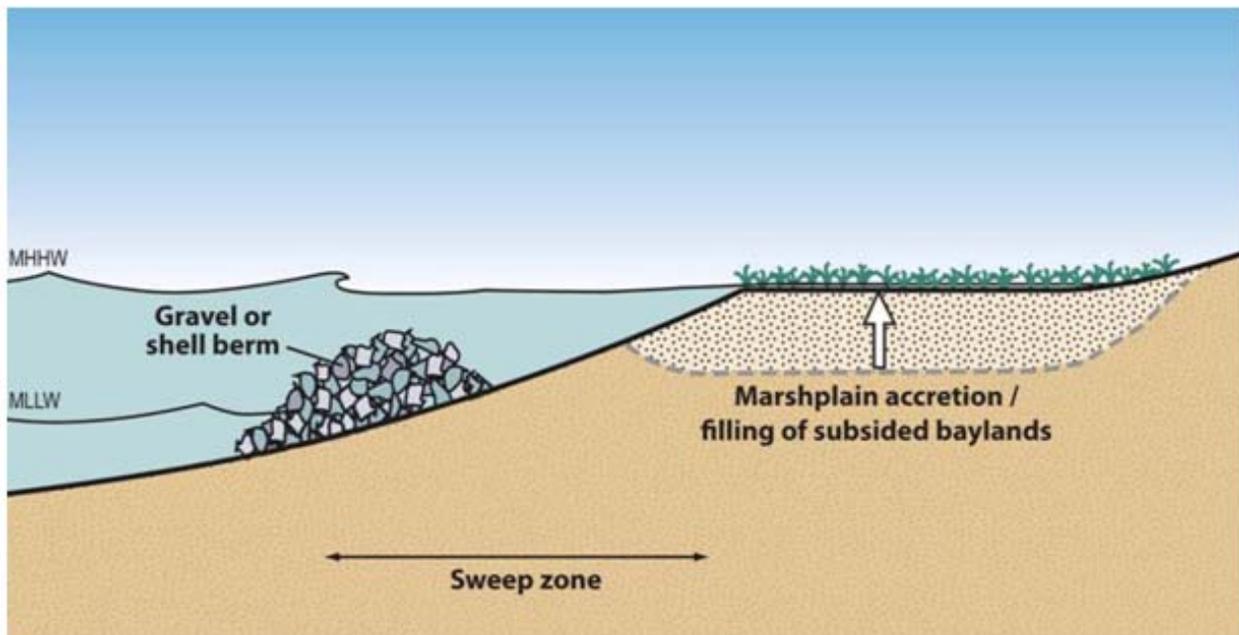
### *Wave Attenuation*

Conventional offshore breakwaters are based on armoring (hardened surfaces, such as rock armor). Low-crested berms constructed from coarse gravel or oyster shell (Figure 2.10), are potential alternatives. These would be able to accommodate rising sea level by naturally rolling landward, driven by wave forces. They may also enhance rather than conflict with ecological and aesthetic objectives for tidal wetlands, and provide additional recreational benefits in suitable locations.

For typical nearshore conditions in the East Bay, the wave heights could be reduced by between 10% and 70% by these types of structures during normal tidal conditions which could significantly reduce horizontal erosion rates. The height of the berm will determine the amount of wave attenuation. The offshore distance of the berm from the marsh edge, together with the length of the berm, will determine how much of the marsh is protected.

### Opportunities and Constraints

Several studies have found that breakwater reefs constructed of loose oyster shell provided substrate for oyster recruitment and harbored a more diverse community of fish and invertebrates than control areas without reefs (Subtidal Goals 2010).



**Figure 2.10** Gravel or shell wave attenuation berm – note vertical exaggeration

While oyster shell breakwaters have been successful in creating valuable habitat, they do not provide the same level of protection that could be offered by traditional engineered designs. However, this approach could serve as an immediate solution to the marsh/shoreline losses experienced along the bay shore.

The potential impacts of offshore wave breaks include smothering of invertebrate communities from both placement of shell and potential accretion in the lee of the structure, local scour around the structures, the possible need for long term maintenance/replenishment, and potential hazard to navigation and recreation.

### Landward Migration

The purpose of the measure is to increase the space available for the landward migration of the tidal marsh with sea-level rise. The intent is to slow the loss of salt marsh due to “coastal squeeze” against steep levee faces, allowing the marsh to maintain its width for longer. More details of the management of the transition zone are provided in Science Foundation Chapter 4.

The rate of landward migration is a function of landward slope and the rate of sea-level rise. Wide transition zones adjacent to high marsh will allow the migration of tidal wetlands with rising sea level, as opposed to being squeezed against steep-sided levees. Healthy transition zones and terrestrial buffers also transport surface and subsurface flows of water and sediment, maintain water quality, provide nutrient input from decaying plants, stabilize shorelines, and store flood waters. All of these functions are affected by the width of the transition zone. These measures are summarized in Table 2.3.

**Table 2.3.** Landward migration management measures

Key Factor	Driver of Change	Measure	Examples
Landward Migration	Landward Elevation and Slope	Create transition zone through placement of fill	Placement of dredged material bayward of levee to increase elevation
		Managed realignment	<ul style="list-style-type: none"> <li>• Levee realignment in a landward location</li> <li>• Relocation of people and infrastructure out of a flood hazard zone</li> </ul>

### *Create Transition Zone*

This measure creates a transition zone on fill slopes landward of the tidal marsh (OLSD 2013). These slopes of 1:20, 1:30 or more could:

- create transition zone habitat that is missing in many parts of the bay due to diking;
- create gently sloping habitat between tidal marshes and existing flood risk management levees to act as buffers to sea-level rise to allow migration using broad, gently sloped gradients;
- provide additional ecosystems services such as act as treatment to polish wastewater discharge.

The transition zone slopes would be an engineered equivalent of lowland floodplain moist grassland habitat (lowland wet grassland and sedge-rush meadows) of broad, flat alluvial fans that historically graded into the tidal marshes of most of South San Francisco Bay. When such slopes have been created in the Bay in the past they have been invaded by pepperweed (*Lepidium*), therefore creating a salinity gradient across the slope would allow native plant species to compete more effectively. For example, treated wastewater could be allowed to seep through the ecotone slope to support the moist grasslands.

The transition zone would be located landward of the existing tidal marsh and bayward of the flood risk management levee (or other logical configurations given the site specific layout). Rather than placing fill directly on existing marshes fronting levees there maybe opportunities to fill manmade ponds (such as salt or oxidation ponds) which lie between the levee and the outboard marsh. Refer to Science Foundation Chapter 4 for a more detailed discussion of transition zones.

### Opportunities and Constraints

This measure would provide long gentle slopes more in keeping with natural historic marsh edge slopes than steep sided levees. The measure will help maintain tidal marsh ecosystem services, such as wave attenuation, for longer with rising sea levels by allowing landward migration. These shallower slopes would allow tidal marshes to migrate landward rather than be squeezed against steeper levee slopes. The now-rare groundwater seep-dependent transition zone would provide seasonal terrestrial habitat for the endangered salt marsh harvest mouse spring foraging habitat and increasingly important terrestrial high tide refuge, particularly as sea-level rises.

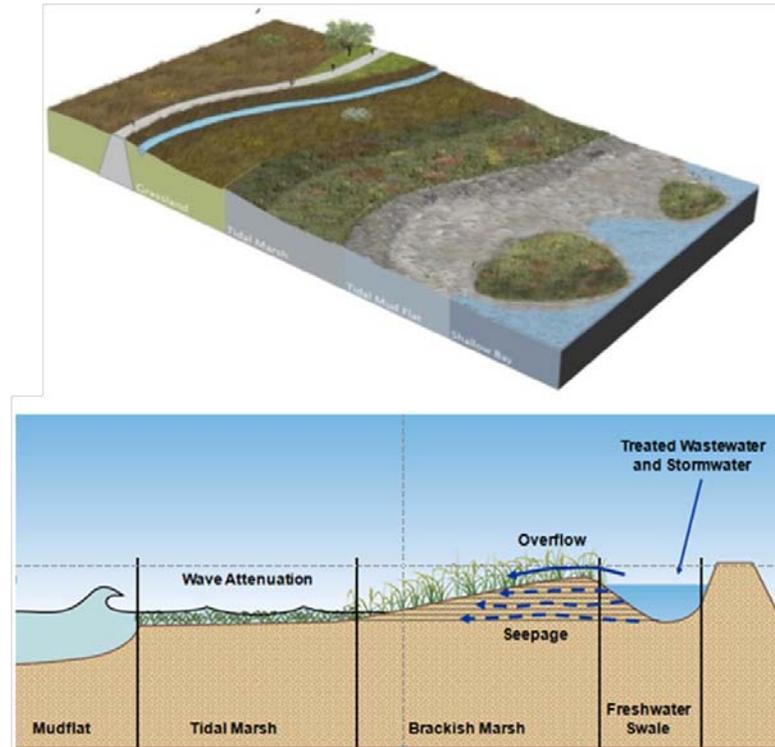


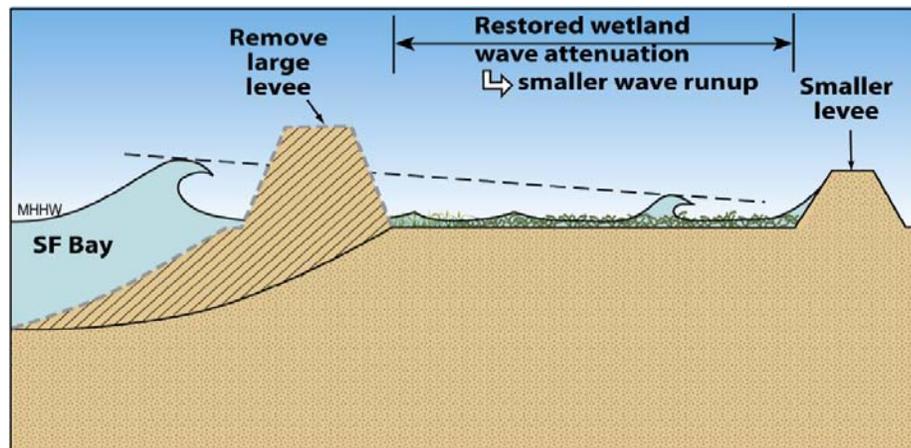
Figure 2.11. Oblique and cross section through transition zone.

However, impacts to existing wetlands could present permitting challenges associated with converting existing tidal marsh habitat to terrestrial habitats and brackish wetlands. In addition, the construction and maintenance of the transition slopes would require large volumes of fill material. Water or land access would be required to allow the placement of fill including up to 3 feet of clean, capping soils to accommodate the rooting zone of the terrestrial native plants. If brackish marshes are to be constructed then a supply of fresh water is required.

### *Managed realignment*

A complementary strategy is to realign the flood risk management levee to a new location further inland. This allows marshes and mudflats to migrate landward naturally. Realignment takes advantage of the natural protection provided by marshes and mudflats (including extant, restored, and potential future marshes and mudflats) to reduce the risk of flooding and erosion allowing smaller levees to be built (Figure 2.12).

Bayland slopes behind the existing levees are often very flat (1:1000 in subsided areas) and tidal marsh accretion rates may not be sufficient to keep up with rising sea levels. This means that the rate of landward migration of the shoreline could be very rapid under certain scenarios. In concert with the moving shoreline, the hazard zone associated with flooding will also move inland. Realignment over relatively flat slopes uses large amounts of land, but may provide flood risk management benefits for only a relatively short period, particularly if vertical accretion rates and plant establishment rates lag behind sea-level rise. It would be necessary to include other measures to promote vertical accretion, such as enhancing channel networks, together with managed realignment.



**Figure 2.12.** Managed realignment of levee to create opportunities to restore tidal action to diked baylands.

### Opportunities and Constraints

Environmental benefits include the potential to mitigate the effects of previous diking and of future sea-level rise, including the potential for new habitats to compensate for change to other habitat of the same type elsewhere. Ultimately, the goal would be the development of a sustainable estuary shape to contribute to flood risk management.

Managed realignment also has flood risk management benefits. These include the potential to change the hydrodynamics of an estuary so as to reduce the risk of flooding at another location or to improve the functioning of the hydrodynamic and sedimentation system. In addition, it can reduce the costs, particularly where it is no longer economic justification to defend land or where realignment enables the levees to be moved to naturally higher ground.

However, managed realignment has some significant capital and political constraints, especially if it requires the relocation of people or critical infrastructure. Areas of sensitive habitat or possible contamination sources that are currently located landward of existing flood risk management structures would also need to be carefully considered.

### *Management Action Strategy*

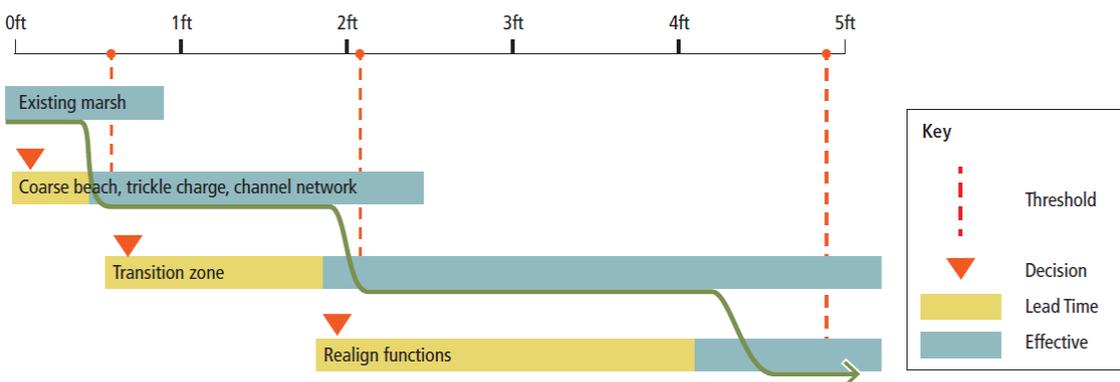
#### Decision Process

The selection of management measures for implementation will require thoughtful consideration of trade-offs between competing uses, near- and long-term benefits and impacts, and different priorities regarding ecosystem services (i.e., which are protected and to what degree). The decision about what constitutes near-term and long-term will depend on the rate of sea-level rise, and in particular when certain threshold elevations will be crossed that trigger the need for intervention (BCDC 2013). Ultimately, successful implementation of a sea-level rise management action strategy will involve adaptive management, defined as a rigorous process of learning by doing and using the results to improve management actions (Figure 2.14). Restoration practitioners have found that, because knowledge of natural and social systems is incomplete, systems can respond in unexpected ways (Trulio 2007). Given this, many data gaps can only be addressed by implementing management measures and conducting long-term monitoring to evaluate their performance.

A distinct conceptual sea-level rise management action strategy will need to be developed for each Segment or distinct marsh unit. The conceptual frameworks will likely consist of multiple management measures to be implemented in a number of phases dependent upon the amount of sea-level rise. The first phase provides immediate ecological benefits to enhance the existing marsh and to maximize its resilience to 2050 – 2070, when sea-level rise rates will still be relatively low. The second phase prepares the marsh for accelerating rates of sea-level rise expected after 2070, when rates may out-pace vertical accretion and marshes will need to migrate landward to survive.

The recommended actions for each baylands segment (main report under ‘New Opportunities: How We Can Achieve Healthy, Resilient Baylands,’) initiate this planning process by providing near- and long-term visions and accompanying actions to take. Actions plans for each marsh can then be built out from the more general segment plans.

Localized conditions will play an important role in the decision making process as to which areas should be selected for action. The decision about when to implement each of these measures will depend on the rate of sea-level rise, and in particular when certain threshold elevations will be crossed that trigger the need for intervention. Figure 2.13 illustrates this concept graphically. Similarly, because of the need to capture as much sediment as possible, areas of the Bay that have high suspended sediment concentrations and a recent history of rapid accretion rates should be prioritized for tidal marsh restoration. Restored marshes with a greater elevation capitol will be more resilient in the long term.



**Figure 2.13.** Potential phasing of adaptation measures triggered by rising sea levels. The timing of different strategies is set by Bay elevation rather than chronological time. Because many of the management measures have fairly long lead times for planning, permitting, and construction, decisions about how and when to implement them will have to be made well in advance of when they are needed.

Finally, because many of the management measures have fairly long lead times for planning, permitting, and construction, decisions about how and when to implement them will have to be made well in advance of when they are needed. For example, large-scale restoration projects can take up to a decade to plan, permit, fund and implement. More challenging projects such as large levees can take longer, and building consensus on innovative concepts, or controversial actions such as managed realignment, could need multiple decades of planning.

### Trade-offs of marshes versus hard engineering solutions

In the 2013 Bay Institute report (TBI 2013), findings indicate that tidal marsh can reduce storm wave heights by over 50% depending on water depth and marsh width. This finding suggests that flood risk

management is improved significantly when areas of tidal marsh exist between the developed shoreline and the open waters of the Bay. Further, by using tidal marsh in combination with a levee constructed at the landward edge of the marsh, the size of the levee could be reduced significantly while still providing the same level of flood protection benefit as would be provided by a larger levee that was not fronted by tidal marsh.

Their analysis concluded that a flood risk management system comprising a landward levee and an adjacent tidal marsh provides an equal level of flood protection to that of a much larger landward levee alone. Moreover, the results indicate that it would be more cost effective to build a flood risk management system that incorporates a tidal marsh than it would to only build a conventional earthen levee.

Significant marsh restoration efforts already are underway in San Francisco Bay. What began with a small, one-off project in the late 1970s has evolved into a regional program with the goal of restoring over 100,000 acres of bay marshes. However, that program has only lately come to incorporate sea-level rise projections into marsh restoration design. Restoration scientists now recognize that many of the restored wetlands are at risk of being drowned by rising tides. In addition, the decreasing availability of suspended sediment in bay waters also poses a threat to the success of marsh restoration efforts.

A new restoration design is needed in order to respond to these changing conditions. The TBI horizontal levee study describes a potential new marsh restoration paradigm that is appropriate in many parts of the Bay and that can provide an interim solution to the problem of tidal marsh inundation and low sediment supply. This new paradigm recommends the addition of a broad estuarine-terrestrial transition zone slope of moist grasslands and/or brackish marshes landward of the existing tidal salt marsh. The transition zone slope would provide both elevation and salinity gradients that would allow the tidal marsh to both move landward and accelerate vertical accretion in order to keep pace with sea-level rise. This new marsh restoration paradigm also proposes the use of clean fill material such as upland sources of construction material or sediment dredged from nearby flood control channels as construction and maintenance material for the transition zone slope. In addition, reclaimed wastewater effluent from existing public water treatment plants along the shore could be used to irrigate the transition zone slope and/or create areas of fresh and brackish marsh.

Many other options for addressing sea-level rise should also be explored, including areas where restored baylands may provide adequate flood protection without new levee construction. Restoring severely subsided landscapes and providing sediment to help sustain naturally accreting marshes may be more cost effective than constructing new levees.

### Near Term Priorities

In the near term, the priority should be focused on 1) enhancing the resilience of existing marshes, 2) expediting the restoration of marshes, and 3) creatively retaining or enhancing the habitat functions of the other Bayland habitats. Within the context of these three priorities, we would emphasize the importance of performing pilot studies to understand the efficacy of various innovative enhancement and restoration techniques to enhance the various processes that increase the resiliency of a marsh (e.g. Figure 2.12). By initiating and monitoring pilot projects for less well-understood measures, lessons about potential opportunities and constraints can be learned, and future implementation can more readily achieve project goals. Each of these three near term priorities results in a specific set of recommended actions.

### **1) Enhancing the Resilience of Existing Marshes**

There are varying degrees of understanding about the efficacy of various short term marsh enhancement measures in San Francisco Bay. For example, restoration practitioners have experience improving sediment pathways in Bay tidal marshes, and there is information available on how to design, build, and monitor tidal channel development. However, using eelgrass and oyster reefs (e.g., ‘living shorelines’) to reduce nearshore wave energy and using coarse beaches to stabilize eroding shorelines are currently in the early testing phases in the Bay. Other measures are untested in the Bay, although some have been tested elsewhere. Potential enhancement measures for existing marshes have been described in the preceding sections and can be summarized as:

- A. Reduce nearshore wave energy
- B. Stabilize with coarse beaches
- C. Recharge mudflats and/or marshes
- D. Improve sediment pathways
- E. Enhance sediment trapping

### **2) Expediting the Restoration of Marshes**

There are large areas of former bayland that are in public ownership and are scheduled for tidal restoration. These areas should be prioritized based on their location in the landscape and likelihood for long-term sustainability.

For example, areas of the bay that have greater suspended sediment concentrations or are at a higher starting elevation (more elevation capital) should be prioritized above areas where marsh establishment and persistence is more doubtful. Restoring significant amount of tidal marsh in the next decade will be of critical importance to capitalize on the existing sediment supply in the Bay as well as the relatively lower current rates of relative sea-level rise. The sooner more tidal marshes can become established in the next 30 years, the better.

In addition, all of the concepts outlined above for enhancing existing marshes should also be applied to newly restored marshes where appropriate. In particular, the creation of broad transition zones and the beneficial re-use of dredge material and upland soils.

### **3) Creatively Retaining or Enhancing the Habitat Functions of Other Bayland Habitats**

Other types of bayland habitats besides tidal marshes, especially diked baylands, provide significant ecosystem services. To sustain their functions and values, it may require changing the management regime, and even consider re-thinking their sustainability. Apart from management adjustments, a managed retreat scenario may be appropriate for some of ponds, requiring relocation or abandonment of ponds in areas of higher threat from sea-level rise. Abandoned ponds could then be converted to other (likely tidal) habitat types dependent upon their elevation and location in the bay. However, pilot studies should be conducted on how to maximize or retain the ecosystem services provided by these other bayland habitat types.

### **Long Term Priorities**

In the longer term it will be necessary to consider other measures, in addition to those listed in the preceding section. Successful pilot studies performed in the near term should be scaled up as appropriate, and regional coordination on multi-purpose projects and spatially appropriate habitat trade-offs should be explored.

**1. Increase transition zone**

This measure creates an estuarine-terrestrial transition zone on fill slopes located landward of the existing tidal marsh and bayward of the flood risk management levee. There may be opportunities to fill man-made ponds (such as salt or oxidation ponds) located between the levee and the outboard marsh to avoid placing fill directly on wetland habitats. Broad transition zone slopes would create a habitat type that is missing in many parts of the Bay due to diking, and provide space to allow for landward migration, buffering the tidal marsh from coastal squeeze between a rising bay and steep levee slopes. Although stated as a long term priority, opportunities to couple these features with expedited tidal marsh restoration in the short term should not be missed.

**2. Realign levees**

Realignment of the flood risk management levee to a location further inland is complementary to the aforementioned transition zone slope measure as it provides additional space for landward migration. Realignment would increase the distance between the bay and shoreline development, allowing for the dissipation of wave energy over distances of several hundred feet or more and allowing the construction of much lower levees inland. It can also be done in a way to provide greater connectivity between upstream watershed processes and the baylands.

A future strategy may then include the following ideas:

- Plan for future marsh configurations: fringing marsh with wide transition zone slopes for landward migration
- Natural, process-based connections to the larger watershed for sustained sediment delivery.
- Concomitant changes in habitat extent/conversion
- Begin accumulating/stockpiling material (either dredge material or upland fill) in strategic locations

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**SUMMARY**

Presently, tidal bayland habitats (marshes and mudflats) are accumulating enough sediment to keep pace with near-term projections of sea-level rise. The future of these habitats depends significantly on the actual rate of sea-level rise and the availability of suspended sediments in the bay to support them. In the near term, it appears that marshes and mudflats throughout the Bay will continue to persist. However, greater uncertainty on their fate occurs as we approach the turn of the next century. Similarly, the functioning of diked baylands will be most impacted as sea-level rise escalates later in the century. Continued monitoring remains key to understanding bayland response to sea-level rise, evaluating ongoing changes, and determining the accuracy of past (and future) marsh modeling efforts.

Specific recommendations are outlined in the main report under ‘New Opportunities: How We Can Achieve Healthy, Resilient Baylands,’ but the primary overarching recommended actions include:

- 1) The need to have a strong sense of the management action timeline, linking planning and subsequent implementation to elevation thresholds to ensure adequate preparation time.

- 2) Immediate implementation of pilot studies to explore and validate many of these recommended measures.
- 3) Flexibility from the regulatory community to allow for new and creative solutions when project objectives include restoring or retaining bayland habitat and function.

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# Science Foundation Chapter 3

## Connections to the Bay

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## INTRODUCTION

The open waters of the Bay (defined here to include San Francisco, San Pablo, and Suisun Bays) link the Baylands to each other, to the major rivers through the Sacramento- San Joaquin Delta, and to the Pacific Ocean. We refer to these links collectively as the Bay Connection. This Connection brings the effects of remote changes in the watershed and the ocean to the Baylands.

This chapter describes the effects of climate and other long-term changes that specifically affect the Bay Connection either through effects of the Bay on the Baylands, or the reverse. Changes within the open water are addressed here only if they are likely to interact with the Baylands. This chapter includes a discussion of the Bay Connection for nutrients, organic matter, and biota; some aspects of sediment movement are touched on here but that topic is addressed more fully in Science Foundation chapter 2. This chapter also provides a partial update to the Subtidal Habitat Goals Report (2010), which laid out recommendations for research, protection and restoration of valued subtidal resources.

Our approach is qualitative, in that the details of many of the interactions are uncertain but will likely involve multiple drivers and a network of responses (Roos 1989). For example, rising sea level will interact with sediment availability, salinity, construction of seawalls and levees, and local wind patterns, which together will affect water clarity, local circulation, and habitat suitability for submerged aquatic vegetation (SAV), intertidal wetland vegetation, and animals. These combined interactions will affect sediment availability for marsh replenishment and maintenance and shoreline protection.

The approach in this chapter is based on the concept of risk, which is the product of probability and consequences. The five scenarios (see New Understanding: the Baylands and Climate Change, Appendix D) bracket ranges of likely future conditions within the estuary for temperature, sea level, freshwater flow, and sediment supply, and include one storm scenario. These trends will likely have substantial consequences for the estuary and therefore impose substantial risks. Other trends may have consequences for the Bay but are less well predicted, or are predicted by some analyses and not by others (Science Foundation chapter 1). Still others are likely to occur as a result of direct human interventions. Some events with a low annual probability may lead to severe consequences, notably an earthquake resulting in the collapse of levees in the

Delta and Suisun Marsh, which would alter the tidal prism and the salinity distribution throughout the estuary.

The probability of a particular outcome is the product of probabilities of each link in the causal chain from the driver of change (e.g., rising sea level) to the final outcome (e.g., loss of eelgrass). In this chapter we focus on the changes and events having high probability or large consequences.

We seek to answer the following questions:

- What are the key concerns among the effects of long-term changes in the Bay on Baylands habitats, or the effects of Baylands on open waters?
- What scientific research or monitoring would be most useful in reducing uncertainty in projections of impacts to the Bay Connection?
- If important negative impacts are expected, are there management actions that could increase the resiliency of the Baylands or open waters?

Case studies (Science Foundation chapter 3, appendix 3.1) have been developed to make the general discussion here and in Science Foundation chapter 5 more specific, particularly with regard to how individual species or groups of species may respond and what might be done about them.

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## **THE BAY CONNECTION TODAY**

The Bay and Baylands are linked dynamically through the movement of water, sediments (Chapter 2, nutrients, organic matter, and organisms). Tidal fluctuations in water level in the ocean produce currents that move water between the ocean and the Bay, and between Bay and Baylands. Freshwater inflow from the major rivers and smaller tributaries alters water level and sets up estuarine salinity gradients. Water flows within the Bay are further modified by atmospheric pressure and wind along the outer coast and, locally within the Bay, wind-driven waves and wind set-up of water level. These links provide mechanisms by which changes in the atmosphere, ocean, and watershed can influence the Bay and thereby the Baylands.

Movement of water also moves sediments, which segregate by size into larger grain sizes in areas of energetic waves and currents, and smaller grain sizes in calmer areas. Nutrients and organic matter are transported by currents, and organisms are transported by currents and their own swimming. Tidal movement of water between the Bay and adjacent marshes oscillate: the water moves in on the flood, mixes within the marsh, and moves and out on the ebb. Dissolved substances, particles, and organisms are carried with the water, providing a mechanism both to nourish the marsh (sediment, nutrients, organic matter, organisms) and to remove materials from the marsh (sediment, organic matter, organisms). By moving from place to place, organisms also transport organic matter, e.g., when they spawn in one area and rear in another.

### **Exchange of Organic Matter and Biota**

Marshes and other shallow areas produce large quantities of organic matter per unit area because of their extensive vegetated surface exposed to sunlight and the continual supply of nutrients from the Bay and from land. Primary productivity within a marsh includes that by the rooted plants, attached algae,

epiphytes, and benthic and planktonic microalgae. Because of high light levels, growth of plants generally exceeds respiration by all organisms within a marsh, resulting in net production of organic matter. Over time this excess production must be either buried or exported to the adjacent Bay and terrestrial areas. This export may occur through tidal exchange between the marsh and the Bay, or through active or passive movement of organisms that feed in the marsh but die outside the marsh.

Export of organic matter from marshes to adjacent estuarine waters was first considered as the "outwelling hypothesis" (Odum 1980, Nixon 1980), by which organic matter that is readily digested by organisms provides an important subsidy to nourish adjacent waters of the estuary. The outwelling hypothesis originated in studies of extensive, rich marshes on the Atlantic coast of the United States. Even there, quantitative demonstrations of the importance of outwelling to estuarine or coastal foodwebs were few (Dame et al. 1986). The difficulty arises from the technical challenge of making enough measurements to quantify a small net flux in a large tidal signal with high variability (Dame et al. 1986) and the numerous modes by which that flux can occur (McKee et al. 2006). In addition, dissolved and particulate organic matter produced by rooted vegetation can consist of material that is resistant to biological breakdown and therefore largely unavailable to estuarine pelagic foodwebs compared to that produced by phytoplankton (Sobczak et al. 2002, 2005). Foodwebs within the diverse marshes of the San Francisco Estuary are supported more by local production of rooted plants and epiphytes than by estuarine phytoplankton (Howe and Simenstad 2011), while pelagic foodwebs rely more on phytoplankton production (Grimaldo et al. 2009).

Benthic and planktonic microalgae (phytoplankton) produced in a marsh are directly available to zooplankton and other consumers. Production by these microalgae can be high in marshes and other shallow areas where light can penetrate to the bottom. Therefore a marsh could export living phytoplankton or their zooplankton consumers to adjacent estuarine waters. However, the flow of plankton to and from a marsh depends on both production and consumption within the marsh, including consumption in shallow waters of phytoplankton by benthic grazers (illustrated for flooded islands by Lopez et al. 2006), or of zooplankton by small fish that seek food and shelter in shallow areas (Kimmerer and McKinnon 1989). Marshes can be simultaneously sinks for some zooplankton (copepods) and areas of aggregation for others such as bottom-oriented larvae (Mazumder et al. 2009). Thus, marshes may act either as net sources or sinks for plankton in the adjacent waters, depending on the availability of habitat for small fish and the degree of colonization by benthic grazers such as clams.

The exact details of the exchange processes depend on the physical configuration of the marsh including residence time of the water and the biological composition, i.e., the kinds and abundance of producers and consumers within the marsh, especially the transient organisms. Few of these aspects have been examined in marshes of the San Francisco Estuary. Long-term studies of the channels of Suisun Marsh have revealed a lot about fish assemblages (e.g., Matern et al. 2002, Feyrer et al. 2003) and jellyfish and some zooplankton (Wintzer et al. 2011, Meek et al. 2013), and some detailed studies of exchange processes have been undertaken (Culberson et al. 2004). A general conclusion from this work is that the channels of Suisun Marsh are largely isolated from the rest of the estuary, presumably because of long residence time, such that the assemblages of species are somewhat distinct and vary separately from those of the nearby open waters. The South Bay Salt Ponds, which began to be reconnected to the tidal action of the Bay in 2006, are highly productive and may export organic matter to nearby estuarine waters, although the form of that export has not been determined (Thebault et al. 2008).

## Use of Marshes by Fish and Crustaceans

A marsh could subsidize or be subsidized by an estuarine foodweb through the movements of fish and crustaceans (both also referred to as nekton, active swimmers that are able to move independently of water currents) into and out of the marsh. Nekton feeding in the marsh but spending substantial time outside the marsh will transport organic matter from the marsh to the open water (Kneib 1997). This can occur in two ways: first, organic matter may be transported through repeated bouts of feeding in the marsh by organisms that otherwise reside in the open waters. Second, nekton may use the marsh as a nursery, rearing to a certain size before leaving the marsh to complete their life cycle in the open water. In both cases there is a two-way influence: biomass flows to the Bay where it may support bay foodwebs, and the transient or rearing fish prey upon marsh organisms, potentially exerting an important control on the size and species of biota resident in the marsh, and thereby on other ecological processes.

The reverse pattern whereby nekton mediate a flux of organic matter into a marsh is probably less common because of the high organic production within marshes. Nevertheless, mysid shrimp moved into a marsh at China Camp where they were apparently consumed, implying a flux into the marsh (Dean et al. 2005).

The species of nekton typically found in marsh channels include marsh resident species and species that are characteristic of open waters (see the case studies for herring and anchovy, appendix 3.1). Resident species, as used here, means those species that are more abundant in marshes than in the open waters; transient species are those that either use the marshes seasonally (Matern et al. 2002) or sporadically, including those that use marshes as nurseries (below). The particular species found in any marsh depends somewhat on the salinity of the water there, although some species such as the non-native Mississippi silversides and striped bass occupy marshes over a range of salinity (Moyle et al. 2013). Resident species include sedentary fish such as sticklebacks, sculpins, and gobies, which feed principally on amphipods and other resident crustaceans in the marsh channels and marsh plain (Visintainer et al. 2006). Transient species include the more abundant species of the open waters, such as northern anchovy, Pacific herring, starry flounder, and striped bass, and migrating species such as salmon.

The South Bay Salt Pond Project has documented large numbers of juvenile fish inside restored salt ponds just a few years after breaching, as well as very high productivity of invertebrates such as shrimp as soon as one year after breaching (Hobbs et al. 2012). This high productivity is apparently exported through consumption by larger predators of the open waters (Hobbs et al. 2012). Pilot subtidal restoration projects have documented reproduction of native oysters, bay shrimp, bay gobies, Dungeness crab, red rock crabs, and nudibranchs (State Coastal Conservancy 2013). This may contribute to food resources for other species that use adjacent marshes, but the extent of this support has not been investigated.

## Nursery Function

Some species of nekton may use marshes as nursery habitat, growing in marshes to a certain size and then migrating out to open water (note that this does not include species that reside in marshes throughout their lives). On the east coast, numerous coastal and estuarine fishes and other nekton, including some that support major fisheries, must spend some part of their juvenile life stages in marshes. These include species such as herring and anchovy (Ayvazian et al. 1992). However, no fish species of the San Francisco Estuary appears to require marshes as nursery habitat. For example, northern anchovy are by far the most common fish in the Bay (see the anchovy case study, appendix 3.1) yet are uncommon in marshes of the

San Francisco Estuary (Visintainer et al. 2006, Gewant and Bollens 2012). Pacific herring use shallow subtidal areas including marsh channels for spawning, laying sticky eggs on surfaces such as rock and vegetation (see the herring case study, appendix 3.1), but young herring rear mainly in the open waters of the estuary. Species such as striped bass and longfin smelt can be abundant in Suisun Marsh but are also common in the nearby open waters at all sizes.

There are several potential reasons for the lack of a life history requiring rearing in marshes. First, the species with this life history may have been lost, or they may have adapted by shifting their rearing to other habitats, as marshes were eliminated from the estuary. Moreover, the relative lack of large estuaries along the Pacific coast compared to the Gulf and Atlantic coasts may have limited the evolution of obligate use of estuaries and marshes as nurseries by west coast species.

Nursery support of the entire estuary may be poor for juvenile Chinook salmon, which grow slowly and move rather quickly through the San Francisco Estuary (MacFarlane 2010). This contrasts with the extensive, long-term use of marshes and shallow waters as nurseries by juvenile salmon in Pacific Northwest estuaries (Miller and Simenstad 1997, Bottom et al. 2005).

There is little published information on the extent to which restoration of marshes will restore nursery function to the marshes, and the species likely to use that function. Intertidal habitat is important to young Dungeness crab in Willapa Bay, WA (Holsman et al. 2003, 2006). Additional intertidal habitat in the saline reaches of San Francisco Bay may provide more habitat for crabs and some other species, but the importance of this habitat for these populations here is unknown.

### Links within the Open Waters of the Estuary

Although the San Francisco Estuary can be considered a single water body (Kimmerer 2004), its size and geomorphic complexity result in considerable heterogeneity. For example, different seasonal and long-term patterns of phytoplankton biomass occur in the Delta, Suisun Bay, and South Bay (Kimmerer 2004, Cloern and Jassby 2010). Thus, most studies of estuarine processes consider only parts of the estuary. The geographic scopes of individual management programs and plans are also usually restricted to particular regions of the estuary; for example, the scope of this document excludes the Delta. Nevertheless, tidal currents, the net flow set up by river flows, the influence of salinity stratification, and the movements of organisms all link the segments of the estuary together.

Links between the Delta and the rest of the estuary are more unidirectional than links between say, San Pablo and Central Bays, because tidal currents are weaker so the net river-derived currents are more important. Still the river-derived current is small (a few percent of tidal currents in dry summers), so tides remain an important mechanism for transporting substances and organisms (e.g., Figure 29 in Kimmerer 2004). Under present conditions, the Delta supplies freshwater that opposes intrusion of ocean salt, nutrients largely from wastewater treatment plants (Dugdale et al. 2007), phytoplankton that subsidizes the low-productivity brackish region of the northern estuary (Kimmerer and Thompson 2014), and zooplankton from freshwater into the brackish region (Kimmerer unpubl.).

The Gulf of the Farallones (GOF) is likewise connected to and not particularly distinct from the marine-influenced Central Bay in terms of biota and physical processes. Exchange between Central Bay and the GOF exports low-salinity water, sediment, and estuarine organic matter and organisms from the estuary while importing coastal sediment, nutrients, organic matter, and organisms into the estuary.

## **FUTURE CHANGE: MECHANISMS AND CONSEQUENCES**

How will long-term change affect Baylands through the Bay Connection? We present here an assessment of likely changes in the Bay Connection, notably those that arise through the movement of water, sediment, substances, and organisms between the Bay and the Baylands. This section focuses on the mechanisms and consequences of change within the Bay's ecology as a result of the drivers enumerated through the future scenarios (New Understanding: the Baylands and Climate Change, Appendix D), irrespective of the likelihood of changes in the drivers. The discussion starts with the physical environment, then examines direct influences on biota, and finally addresses biological interactions between Bay and Baylands and within the Bay. Case studies (appendix 3.1) furnish specific examples of the vulnerabilities and potential responses of several species groups and species, selected for their importance or likely responses.

### **Effects on Bathymetry and Sediment Supply**

The bathymetry of the estuary exerts a strong control on circulation of water and sediments, and bathymetry is altered by erosion and deposition of sediments. Bathymetry in turn sets up habitat characteristics throughout the Bay and Baylands, and sediment supply can constrain the formation of Baylands newly connected to the Bay. Wind, biological activity, and human activities around the estuary also influence bathymetry locally.

*Rising sea level* will have numerous effects on Baylands, some of them direct (see Science Foundation chapters 1 and 2, and some indirect through the Bay Connection. The bathymetry of shallow areas of the Bay is in rough equilibrium between sediment supply and erosion due to wind waves and tidal currents. Therefore, the depth profiles of the future estuary will likely be similar to those today in areas where there is sufficient sediment supply and deeper in other areas because of higher sea level. The process of morphologic adaptation may lag sea level rise, producing temporary increases in water depth. One consequence of deeper channels is a greater tendency for stratification to develop, increasing salinity penetration into the estuary (Monismith et al. 2002).

*Sediment availability* depends on sediment supply as well as wind waves, currents, and local factors such as dredging and sediment stockpiles for nourishing wetlands (Science Foundation chapters 1 and 2. Several long-term trends (decreasing sediment supply, rising sea-level, increasing wind waves, and possibly an increase in vessel wakes and construction of seawalls) will combine to reduce the total sediment available within the estuary, although redistribution through these mechanisms may increase it locally. Sediment availability throughout the estuary will also depend on the extent of sediment trapping in restoration sites and in subsided islands in the Delta and Suisun Marsh that are reconnected to the estuary by catastrophic flooding, which may be more probable with higher sea level (Mount and Twiss 2005). The planned replumbing of the Delta includes two large tunnels designed to divert water from the Sacramento River during high flows, and with it about 8-9% of the annual sediment load down the river, in addition to extensive restoration of shallow habitat, both of which will reduce sediment loading from the Delta to the rest of the estuary (BDCP 2013).

Any increase in *wind speed* over shallow parts of the Bay will increase resuspension, which may increase or decrease sediment supply to marshes depending on local circulation patterns and water depth. An increased frequency of major storms would have a similar effect.

*Water clarity* depends on suspended sediment concentrations near the surface, which in turn is limited by the availability of erodible sediment (Schoellhamer 2011). Water clarity at any location is a complex

function of sediment availability, local bathymetry, salinity distributions, current speed and direction, and wind waves. The waters of the Delta and Suisun Bay have become clearer over the last several decades (Kimmerer 2004), and suspended sediment concentrations have decreased in the last two decades (Schoellhamer 2011). These trends are likely to continue.

*Contaminants* can be released when sediments are resuspended through erosion or dredging. This is especially true for sediments deposited during the Gold Rush, which contain contaminants such as mercury and PCBs (McKee et al. 2006). Contamination in estuarine fishes such as striped bass has been attributed to concentration of contaminants such as mercury in the foodweb, and has led to recommended restrictions on human consumption of these species (Davis et al. 2012).

*SAV beds* (see the case study, appendix 3.1) interact with sediment supply. Their maximum depth is limited by light penetration and therefore turbidity, but they also trap and stabilize sediments. As with marshes, SAV beds can presumably migrate up-slope as sea level rises, but that depends on local bathymetry and wave energy. The seaward limit of SAV beds is generally set by light availability, although this limit may become deeper by a decrease in turbidity, favoring more extensive SAV beds.

### Other Effects Propagating from the Ocean

*Upwelling* brings cool, nutrient-rich, low-oxygen, and low-pH water to the surface and promotes phytoplankton blooms in the coastal ocean. Estimates of recent climate-related trends in upwelling and projections of future upwelling have been equivocal. A recent meta-analysis of decadal-scale studies showed increasing intensity of upwelling-favorable winds along the California coast (Sydeman et al. 2014). This could increase the nutrient supply for plants and algae in the estuary. It can also bring in large numbers of diatoms and other plankton that thrive in upwelled waters (Cloern and Dufford 2005), as well as low-oxygen and low-pH (see below) waters. Low-oxygen events associated with pulses of upwelled water have been observed in South San Francisco Bay since 2006 (J. Cloern, pers. comm. to W. Kimmerer, 22 February 2015). A large region of low oxygen centered off Oregon has been expanding (Pierce et al. 2012) and may increase the frequency of hypoxic events in the Bay.

*Ocean climate* refers to the phase of various cyclical patterns of temperature and upwelling such as ENSO, PDO, and NPGO. These shifts alter regional weather patterns, resulting in significant effects on the estuary arising mainly through changes in runoff and, to a lesser extent, temperature. We discuss effects on estuarine species composition below.

*Ocean pH* is decreasing, a consequence of the rise in dissolved CO<sub>2</sub>. The reduced pH will affect the estuary through mixing of relatively acidic ocean water into the estuary. The principal concern for the ocean is the vulnerability of calcifying organisms such as bivalves (Fabry et al. 2008), although not all organisms are likely to respond strongly to acidification (Hendriks et al. 2010). The effect of acidification within the estuary is complicated by high short-term and small-scale variability: the pH of ocean water is affected by upwelling of acidic water (Feely et al. 2008), and the pH of the estuary is affected by plant production cycles, land drainage, and wastewater discharge, which can be high in dissolved inorganic carbon (Feely et al. 2010, Fuller 2010, Cai et al. 2011). The key uncertainty is whether the overall range of pH in the estuary will shift enough to affect biota. Any persistent decrease in pH is likely to impair calcifying organisms, notably native oysters, which may be sensitive in the larval stages (see the oyster beds case study, appendix 3.1). Other biota may be affected as well, but that effect is likely to be smaller and is less certain than effects on bivalves.

Most estuaries, particularly in developed regions, are net consumers rather than producers of organic matter and therefore sources of CO<sub>2</sub> to the atmosphere (Smith and Hollibaugh 1993), although net metabolism in the San Francisco Estuary overall appears to be nearly balanced (Smith and Hollibaugh 2006). Low and variable pH in Puget Sound was due to a combination of acidification and eutrophication (Feely et al. 2010). The waters of Central San Francisco Bay may behave similarly, although pH in the rest of the estuary may be controlled more by local and watershed processes. Monitoring data from San Pablo Bay to the Delta show no temporal trend in pH, but pH decreases and becomes more variable from west (saltier) to east (fresher), presumably because of the buffering capacity of seawater.

### Freshwater Flow, Temperature, and Salinity

**Delta outflow** controls the long-term salinity distribution (Science Foundation chapter 2). It also directly affects the estuary by supplying organic matter, nutrients, and organisms from the Central Valley watershed to the estuary. Delta outflow is positively correlated with the abundance of several key populations of fish and crustaceans in the northern estuary, notably longfin smelt and striped bass (see the longfin smelt case study, appendix 3.1). Therefore reductions in springtime freshwater flow can be expected to reduce abundance of these fishes both in the open waters and within the marshes.

The supply of materials such as nutrients and organisms from the rivers is important to the estuarine foodweb and helps to supplement a depression in summer-fall phytoplankton productivity at intermediate salinity (Kimmerer and Thompson 2014). However, the magnitude of this subsidy is unknown, and the likely direct impact of the projected changes in Delta outflow seem small since summer-fall is usually a low-flow period anyway. The link between any change in this foodweb subsidy and the Baylands therefore seems rather weak.

**Temperature** of the Bay's waters has a spatial gradient that reverses seasonally and interannually. Temperature fluctuation near the Golden Gate is dampened by cool summer climate and low seasonal variability in the ocean (Kimmerer 2004). Air temperature in the Central Valley fluctuates seasonally much more than along the coast, and water temperature follows suit. Thus, the estuary is warmest in the Delta during summer (annual range 10-22 °C at Rio Vista), and warmest at the Golden Gate (annual range 12-15 °C) during winter. Freshwater flow has only a minor effect on water temperature in the estuary (Kimmerer 2004, Wagner et al. 2011).

Projections of water temperature under climate scenarios 1-4 were provided only for the Delta because of the known sensitivity of the endangered delta smelt to temperature above 25°C (see the delta smelt case study, appendix 3.1; Cloern et al. 2007). The number of days of temperatures above 25°C at Rio Vista is projected to increase from zero during the historical period of 1984-1999 to ~100 days per year by 2090 under a business-as-usual scenario (Intergovernmental Panel on Climate Change scenario GFDL-A2) and ~18-20 days per year under the more optimistic PCM-B1 scenario (Cloern et al. 2007, Wagner et al. 2011). The increase in projected maximum summer temperature by the end of this century is about 6 and 3 °C respectively under the two scenarios, and increases in projected winter temperature are about half that much (Wagner et al. 2011). This trend is sharply reduced to the west of the Delta as far as Carquinez Strait (Wagner et al. 2011) and presumably even more reduced in San Pablo and San Francisco Bays.

The daily-averaged horizontal **salinity** distribution of the estuary is controlled largely by Delta outflow, but this control is modulated by bathymetry (previous section, see also Kimmerer et al. 2013). It is important to note that an estuarine salinity gradient is always present from freshwater to seawater, but its position changes with freshwater flow and tides, and it often extends into the Gulf of the Farallones. Shallow

regions of the estuary are usually well-mixed vertically, while deeper regions of intermediate salinity can be strongly stratified, with higher salinity at the bottom than at the surface. The position of the winter salinity gradient is governed mainly by storm intensity and frequency, modulated by the extent of water storage in snowpack (decreasing) and reservoirs (probably increasing). The position of the summer salinity gradient is controlled mainly by operations of the large water projects in the Delta and salinity in some parts of the estuary can be affected by local wastewater discharges.

Forecasts for future changes in salinity patterns are complicated by uncertainties about future water management and the future physical configuration of the estuary (see salinity discussion, Science Foundation chapter 2). Winter salinity patterns may be more variable between and within years if storms become more intense, but this is difficult to predict and would be altered by levee failures in the Delta. Salinity is likely to penetrate further and more persistently into the estuary during the dry season, a consequence of reduced spring-summer runoff, altered structures and operations of Central Valley water projects to conserve water for human use in summer, an increase in tidally linked areas due to restoration and levee failures, and higher sea level. Without a major change in bathymetry or tidal prism, further salinity penetration in summer will mean that the extent of the tidal freshwater reach in the Delta will shrink while the extent of saline water ( $> 20$ ) in the Bay will expand.

Salinity has numerous effects on estuarine physics, chemistry, and biota. Physical and chemical effects are discussed in Science Foundation Chapter 2. Pelagic organisms move with the water and are not greatly affected by changes in salinity, although their distributions move as the salinity field moves. Benthic organisms and attached plants (SAV and marsh plants) can be strongly affected by the salinity distribution, dying back in areas where salinity has become unsuitable and colonizing newly suitable areas. These changes can be rapid, as in the case of native oysters (see oyster bed case study, appendix 3.1) which died back following high-flow periods in 2006 and 2011 and subsequently recolonized areas in northern Central Bay. Eastern oysters have thrived in reduced salinity because it provided a refuge from less tolerant disease organisms (Hoffmann et al. 2009, Levinton et al. 2011).

Die-backs of eelgrass and possibly other organisms during winter may affect Baylands through long-term shifts in distribution and accompanying shifts in sediment trapping capacity. Higher average salinity in the future may allow eelgrass, native oysters, and other salt-tolerant benthic or marsh organisms to colonize further up the estuary.

### Extreme Events

Under the storm scenario (New Understanding: the Baylands and Climate Change, Appendix D), extreme winter flow combined with sea level elevated by the long-term trend combined with an El Niño would cause extensive flooding and drive the salinity field far to seaward of its usual winter position. The spring 1986 flood, upon which this scenario was based, kept Suisun Bay fresh for about two months. With higher sea level at the Golden Gate the response of salinity to flow would be somewhat reduced. The principal effect through the Bay Connection would be this suppression of salinity, which would cause temporary die-backs of some species and range extensions of freshwater species as discussed in the previous section, and could set up conditions favorable for invasive species to establish, as may have been the case with the clam *Potamocorbula amurensis* (see the plankton case study, appendix 3.1).

A winter flood could also have lasting consequences through rearrangements of the sediment distribution (Science Foundation Chapter 2) or if flooding breached levees in the Delta or Suisun Marsh and reconnected present-day lands to the estuary, and a decision were made not to repair the breaches. Such a

decision seems more likely in the event of massive levee failures resulting from an earthquake, because repair would take considerable time and may not be feasible for all levees (Mount and Twiss 2005). Over the time scale of a century, these events have a high probability of occurring, but the probability of a permanent response in the estuarine biota that affects the Baylands is highly uncertain.

### Warming Effects on Biota

**Water temperature** affects estuarine biota in several ways, which will vary seasonally and spatially, because of the gradient in temperature discussed above. Temperature sets the biochemical rates that determine the physiology of most estuarine organisms. Increasing temperature will generally increase metabolic rates, except for species near their upper thermal limits which will begin to suffer ill effects. The likelihood of such effects increases with distance into the estuary because of the moderating effect of cool coastal conditions (see Temperature, above). However, there may be some capacity for populations to adapt to altered temperatures.

Rising temperature may stimulate an increase in incidence of disease and parasite attack, although there are almost no data on current levels in any estuarine organisms (but see Friedman et al. 2005 for native oysters). Blooms of the freshwater microalga *Microcystis* occur in the Delta during warm, dry summers and may increase in duration with warming. Higher winter temperatures or a shorter winter season may affect phenology or reproductive success of some species, and provide more favorable conditions for species intolerant of low temperature, although winter temperatures in the estuary are already mild.

Ecological interactions (discussed further below) will amplify the species-specific responses to temperature which will make changes in the ecosystem difficult to predict. These may include changes in phenology that put organisms out of phase seasonally with their food or predators, and changes in frequency or severity of outbreaks of disease or parasites. Overall we can expect some species to be extirpated, some to decrease in abundance, others to increase, others to change seasonal patterns, and still others to extend their ranges into the estuary and become established. The outcome will be an unpredictable shift in the composition of the estuarine biota.

A few species may already be near their upper thermal limits and further increases are likely to prove harmful. In particular, high summer temperature in the Delta will add to the problems already besetting delta smelt, although the link to Baylands is weak (see the delta smelt case study, appendix 3.1).

High summer air temperature, stronger wind, and a greater tidal range may increase the risk of desiccation in intertidal areas (see the rocky intertidal organisms case study, appendix 3.1). Rising sea level is likely to shift the distributions of intertidal organisms higher where space is available, but should not alter the risk of desiccation since the entire assemblage would presumably maintain its position relative to the tidal range.

**Remote changes** refer to events happening remotely from the estuary that influence the estuary through migratory biota. Such effects are particularly likely for birds that spend summers in the Arctic (Science Foundation chapter 5) and for salmon which will encounter increases in river temperature and reduced summer flow rates. In particular, high water temperature in Central Valley streams, particularly in combination with low flows in the dry season and a limited cold-water pool in the reservoirs, are likely to limit the viability of some salmon runs, notably winter-run Chinook (see the salmon case study, appendix 3.1). The loss or reduction in abundance of salmon in the estuary during the outmigration period may have ecological effects on the Bay Connection, but these cannot be predicted.

### Estuarine Productivity

Productivity of estuarine waters depends largely on phytoplankton (microalgae) rather than external sources such as rivers or marshes (see the plankton case study, appendix 3.1). Benthic microalgae may be an important source in shallow areas but their contribution to estuarine productivity has not been studied. SAV beds are an important source of organic matter locally, but their limited extent constrains their contribution to overall estuarine productivity (see the SAV case study, appendix 3.1). Therefore the key controls on phytoplankton set important limits to estuarine productivity. These controls are light, nutrients, and grazing by clams and zooplankton. Similar controls, plus availability of suitable sediment, limit the productivity of benthic microalgae and SAV beds. All of these controls are likely to have trends through time (see the plankton case study, appendix 3.1).

Light available to phytoplankton is a function of sunlight, water depth, and turbidity of the water. Mean depth of the estuary is expected to continue increasing with rising sea level and erosion. However, the main control on light availability is turbidity, a function of suspended sediment concentration (see above). Turbidity has been the key factor preventing the estuary from becoming eutrophic, i.e., productive enough to cause problems with low oxygen and nuisance blooms of algae. There is concern that some parts of the estuary may become eutrophic once turbidity decreases through reduction of net availability of sediment. This trend could be offset to some extent through nutrient sequestration in marshes if the total area of marshes increases.

Nutrient loading increases with human population size, but loading of some nutrients may be reduced, despite population growth, because of upgrades to water treatment plants. Since the estuary is already nutrient-rich, the overall reduction will mainly limit the maximum biomass of phytoplankton blooms in South to San Pablo Bays, and possibly growth of SAV, its epiphytes, and macroalgae. Reduction in ammonium loading may allow for higher phytoplankton production in Suisun Bay (Dugdale et al. 2007), but the likely magnitude of such an increase is controversial because of other strong controls on phytoplankton production, notably grazing by clams and microzooplankton (Kimmerer and Thompson 2014).

Grazing by clams has imposed a severe limit on phytoplankton biomass in Suisun Bay since the arrival of *Potamocorbula amurensis* in 1987. Species composition of the phytoplankton in the northern estuary has shifted toward smaller forms and more toxic or less nutritious forms (see the plankton case study, appendix 3.1).

Phytoplankton biomass (as chlorophyll) has increased markedly in South Bay over the last 15 years, presumably because of a reduction in grazing by clams. This decrease was attributed to a shift in the ocean climate, resulting in a shift in the species composition of the predators in the coastal ocean and a consequent increase in predation on clams (see the plankton case study, appendix 3.1).

### Species Composition

Species composition of estuarine flora and fauna varies as a function of the salinity and depth distribution, seasonal patterns of abundance, and interactions among species. Species composition is likely to change in ways that influence the Bay Connection (see case studies for all estuarine species, appendix 3.1). This change can arise through four mechanisms. First, new species will almost certainly be introduced either accidentally or deliberately. Second, new species may enter the estuary through range expansion that occurs with changing conditions in the ocean or in remote habitats (see previous section). Third, habitat

suitability for different species within the estuary may change through any of the mechanisms discussed above, to the extent that species are extirpated or their abundance or ranges within the estuary are altered. Finally, interactions among species are likely to cause long-term changes in species composition (see below). These changes could be catastrophic or beneficial, and may or may not affect the Bay connection; furthermore, most of the changes are unpredictable.

Disruptive **species introductions** in the past provide excellent case studies for projecting future trajectories of change resulting in and from changes in species composition. The San Francisco Estuary is known for the high level of introduced species in its flora and fauna. Two aspects of species introductions are relevant here. First, the invasion event can disrupt the extant ecosystem as described in two examples below. Second, established introduced species become part of the estuarine ecosystem, but their resource use may come at the expense of native species. Although some introduced species are favored for their value in fisheries (striped bass, largemouth bass) and other human activities, there is also considerable interest in protecting and maintaining some native species, notably cordgrass, eelgrass, oyster habitats and delta and longfin smelt (see case studies, appendix 3.1).

The best-studied introduction event was that of the “overbite” clam *Potamocorbula amurensis*, which has caused suppression of phytoplankton and zooplankton from San Pablo Bay to the western Delta since 1987 (see the plankton case study, appendix 3.1). Although variable salinity patterns likely fostered the settlement of this clam (Nichols et al. 1990), its source was most likely ballast water, and such introductions can generally be seen as chance events. This event has reverberated through the foodweb of the northern estuary, where numerous populations of fish are in a state of decline, to which the chronically low abundance of food is a likely contributing factor (Sommer et al. 2007).

Another disruptive introduction was the Brazilian waterweed *Egeria densa*, which spread throughout the freshwater regions of the Delta in the 1990s (SAV case study, appendix 3.1). This species provides habitat for a host of non-native fishes and other species, and effectively excludes native species of fish and other SAV. Its spread is the principal reason why the Delta is now a favored spot for tournament fishing on largemouth bass, an introduced predator. Although *E. densa* is not abundant west of the Delta, its effects on the estuarine foodweb are considerable and very likely influence the Bay Connection in Suisun and San Pablo Bays.

Future introductions are likely to have effects of similar magnitude, although generally it is difficult to anticipate what species might arrive here. Key exceptions are quagga and zebra mussels, which are both already established in California reservoirs and are readily transported on trailered boats. Although these are freshwater mussels and therefore unlikely to become very abundant in the Bay, they could have substantial effects through grazing on phytoplankton in the Delta. Under present conditions the Delta subsidizes phytoplankton in Suisun Bay; therefore a loss of productivity in the Delta could have major impacts on Suisun Bay (plankton case study, appendix 3.1). In addition, loss of productivity in the Delta would reduce food available to species such as striped bass, which is abundant throughout the estuary and uses Suisun and other marshes (Matern et al. 2002).

Changes in the geographic range of marine species track changes in the geographic distribution of temperature (Pinsky et al. 2013). This can lead to changes in the composition of marine species at the entrance to the estuary, we are aware of no examples of permanent range extensions into the subtidal waters of the estuary. The influx of predators on clams discussed above was apparently a result of decadal-scale oscillations in ocean climate (Cloern et al. 2007). The recent return of harbor porpoises to the Bay is also a range extension, although the reasons for their return after 65 years’ absence are unknown

([http://www.ggcetacean.org/Harbor\\_Porpoise.html](http://www.ggcetacean.org/Harbor_Porpoise.html)). The suite of species that could enter the estuary from the ocean changes with ocean climate on decadal and seasonal scales, so it may be difficult to detect a long-term change in range of these species.

Another mechanism influencing species composition is differential responses of species to changes, especially in temperature and salinity. Rising temperature will affect seasonal patterns, more strongly in inland areas. Seasonal patterns are likely to shift as heat-tolerant species extend their period of high abundance, and heat-intolerant species retreat (if possible) to a winter peak in abundance. These changes will be influenced by shifts in phenology, such as earlier reproduction by species keyed to the spring temperature increase.

Landward shifts in the salinity field of the estuary will affect spatial distributions, more strongly in the summer dry season. As the dry-season salinity field shifts landward, the geographic distribution of these species will also shift. Changes in response to salinity are clearly evident in spatial shifts of various species of pelagic fish and plankton, which maintain positions more in relation to salinity than to geography. The region of overlap in range of the clams *Potamocorbula amurensis* and *Corbicula fluminea* shifts with the limit of tidally-averaged salinity penetration into the estuary (Brown et al. 2014).

Some species with limited geographic distribution may be strongly affected by shifts in salinity. Examples include native oysters, eelgrass, and Pacific herring (see case studies, appendix 3.1). These species require suitable substrate for attachment, bed development, and spawning, respectively, which is most available in and near Central to southern San Pablo Bays, and less so further up the estuary. Increasing salinity penetration may expand the range of eelgrass but the ranges of oysters and herring may not expand because of the lack of substrate. In particular, herring require some dilution of seawater for successful reproduction, and may not be able to expand their range because of the shortage of substrate.

### Consequences for the Bay Connection

The result of shifts in species composition within the open waters of the estuary will be a change in the suite of species available for interactions within marshes. This effect will be most prominent in brackish marshes such as Suisun Marsh, where the scope for changes in salinity is the greatest. Presumably some marine species will be able to penetrate further into the estuary and become significant members of the marsh fauna in areas where they are not now abundant.

Estuarine nekton can exert a controlling influence on species composition and processes in marshes through predation on marsh-resident organisms (Kneib 1997). Therefore changes in the species composition of the estuarine nekton could have a substantial effect on ecological processes with the marshes. By the same token, changes within the marshes could lead to greater or lesser use of the marshes by Bay nekton or alter the biological links between marshes and open waters in other ways. The principal mechanisms for such changes will be the net increase in marsh area due to restoration and protection of marshes from erosion (Chapter 2, and changes in species composition within marshes due to introductions or changes in range (Science Foundation chapter 5).

As with estuaries more generally, a key unknown is the extent to which marshes, either in their current state or in a future state of development, support or would support Bay foodwebs through export of organic carbon or organisms. The estuarine foodwebs in Suisun Bay and the Delta are supported largely by local phytoplankton production rather than inputs from rivers and marshes (Sobczak et al. 2002, 2005), and stable isotope studies show that open-water foodwebs gain little organic matter from marshes or nearshore

vegetated areas (Grimaldo et al. 2009). At present the area of extant and restored tidal marsh is about 18% of the area of the estuary, both excluding the Delta (Kimmerer 2004). Marshes are much more productive per unit area than open waters, but only a small fraction of that productivity is available to support open-water foodwebs. Thus, under current conditions the limited extent of marshes surrounding the estuary suggests a small contribution to estuarine foodwebs baywide. Increased marsh development may increase the magnitude of that subsidy but its importance remains to be determined. This highlights the need for better information on the Bay Connection and for research to accompany all restoration programs.

### Interactions

The above discussion shows the importance of interactions among the links comprising the Bay Connection. Ecological interactions are arguably the least predictable but among the most important mechanisms for changes in species composition in the estuary. These interactions could take many forms, including predation, competition, disease, and parasitism. Examples of such interactions abound in the literature on terrestrial systems, although estuarine examples are harder to find.

A handful of examples from the San Francisco Estuary show the importance of interactions in setting the spatial and seasonal distributions of species, but also in the difficulty of detecting such interactions. The trophic cascade initiated by a shift in ocean conditions, resulting in an increase in phytoplankton biomass in South Bay, is discussed above (see also plankton case study, appendix 3.1).

The introduction of the clam *Potamocorbula amurensis* precipitated a series of events including a decrease in phytoplankton production and a substantial contraction of the salinity range of northern anchovy, which had been the most abundant fish as far up-estuary as Suisun Bay (plankton and anchovy case studies, appendix 3.1). Decreases in spring-summer abundance of several species of copepod and mysid had strong effects on the availability of food for fish, which probably caused a decline in abundance of longfin smelt and striped bass. This complete rearrangement of the trophic links in the brackish parts of the estuary was the most prominent example of a complex network of ecological interactions ever observed in the estuary.

Abundance of *P. amurensis* in San Pablo Bay decreases each fall-winter, apparently because of predation by waterfowl (Poulton et al. 2002). This results in an annual pattern of abundance of the clam, and the possibility of spring phytoplankton blooms occurring before the annual increase in abundance of clams (plankton case study, appendix 3.1). This pattern depends on the continued high consumption rate by the waterfowl, which may be under the control of processes in their distant summer habitats.

A hybrid of native cordgrass with the invasive cordgrass *Spartina alterniflora* has colonized estuarine mudflats throughout saline parts of the bay. Changes in physical and chemical characteristics of the sediment in the colonized areas have led to reductions in survival and abundance of many foodweb organisms that are important to consumers such as birds (Neira et al. 2006).

The interactions discussed above were all associated with rapid changes identified in monitoring programs that included many of the key species involved. Trends in the key drivers of change are likely to be much slower. Thus, ecological interactions are often subtle, complex, difficult to detect or identify, and nearly impossible to predict (Figure 3.1). They will likely amplify or override some of the more direct effects discussed above. Therefore it would be unwise to attempt more than a broad, general forecast of changes in species composition except for the rather straightforward effects of salinity and effects on individual, well-studied species.

The importance of ecological interactions is illustrated in Figure 3.1. Each species responds in its own way to aspects of its physical environment such as temperature, salinity, turbidity, and the available of physical habitat (e.g., water depth and velocity, sediment availability and type, bathymetric complexity). Each species also interacts with a variety of other species where they overlap in time and space. Interactions include consumption of one species by another, providing an energy source to the consumer or parasite but imposing mortality on the prey, which can also impose constraints on prey behavior and selective pressure on prey attributes such as size at maturity.

## INFORMATION NEEDS AND KEY UNKNOWNNS

Key information needs for the Bay Connection were identified in the SHGR. The fundamental unknowns for identifying, forecasting, and responding to changes in the Bay arise from uncertainties in the drivers, which are discussed in Chapter 1 and 2.

Some unknowns could be reduced to clarify and anticipate the likely response of the estuary to long-term change. The specific characteristics of exchange between marshes and open waters are key to understanding the Bay Connection, but are very poorly understood. For example, we have no estimates of organic matter leaving marshes anywhere in the system. Except for a study showing net consumption of mysids in the China Camp marsh (Dean et al. 2005), we do not know the sign or magnitude of the fluxes of phytoplankton or zooplankton between marshes and open water. We do not know how important the

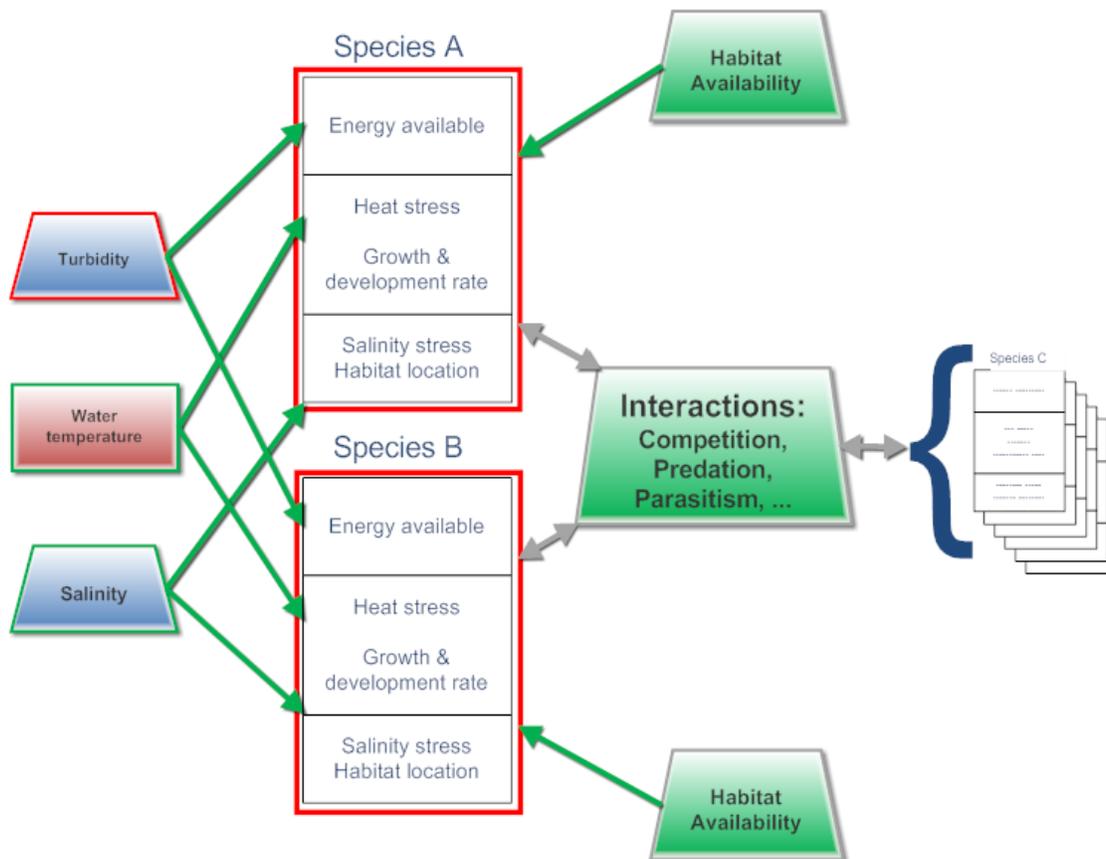


Figure 3.1. Simple representation of ecological interactions and how they effect local abundance of a species.

marshes are to the populations of fish and crustaceans that use them, or the extent to which organic matter produced in the marshes is carried out into the Bay by nekton. These all point to a program of investigations to determine these fluxes and how they may change in the future.

Understanding of the ecology of the San Francisco Estuary has benefited from data provided by two long-term monitoring programs that measure phytoplankton biomass and basic water quality (temperature, salinity, dissolved oxygen, various measures of turbidity). The Interagency Ecological Program (IEP) samples monthly for these variables from the Delta to San Pablo Bay. The US Geological Survey (USGS) monitors monthly in the channels from South Bay to Rio Vista in the western Delta. Both programs maintain continuous monitoring stations that measure temperature, conductivity (salinity), and in some cases dissolved oxygen, suspended sediment concentrations, and other variables. The IEP also samples monthly for zooplankton and macrobenthos at many of the stations sampled for phytoplankton. Fish and other nekton are sampled by several programs, most of which emphasize the Delta and Suisun Bay. The IEP's San Francisco Bay Study samples for nekton throughout the estuary although with limited coverage of the Delta. The data produced by these programs have been critical for investigating causes of declines in fish abundance and other changes (e.g., Thomson et al. 2010, Mac Nally et al. 2011). Finally, the Regional Monitoring Program (RMP) samples throughout the estuary semiannually for contaminants and benthos (Anderson et al. 2007).

Despite this extensive monitoring, substantial gaps exist in the information available to support decision-making. None of these monitoring efforts samples for zooplankton in Central and South Bay, despite their key role in estuarine foodwebs. Sampling for benthos is inadequate to detect trends in abundance or to provide estimates of grazing rates on the overlying phytoplankton, particularly in Central and South Bay. Sampling for phytoplankton includes taxonomic composition but biomass estimates include only bulk chlorophyll, and there is no sampling at all for microzooplankton, which at times are the predominant grazers on phytoplankton. There is no monitoring program that adequately samples jellyfish, which may have increased in harbors in the estuary and perhaps also in open waters with potentially large effects on the estuarine plankton (Mills and Rees 2000). Most of the monitoring focuses on abundance and species composition, and there is no monitoring of vital rates, which would be essential for identifying trends in response to temperature or ecological effects.

IEP monitoring is mandated by the State Water Board, has a funding stream, and is likely to persist, although efforts to expand the scope of the zooplankton program in 1997-1999 did not go beyond the pilot stage. The USGS monitoring program is not mandated, despite its immense value in tracking and understanding changes in phytoplankton and water quality within the estuary, and the publication of ~100 papers, some very influential, using data collected in this study.

It is important to note that monitoring is essential for detecting trends and changes, but monitoring is insufficient to determine cause-effect relationships or linkages within the ecosystem. Therefore, even if the gaps in monitoring were to be filled, our understanding of system structure and function would be inadequate without a research program targeted at the key unknowns.

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## **ADAPTATION**

Although the high level of uncertainty makes planning difficult, the Bay Connection offers some opportunities for adaptation to long-term change. The Subtidal Habitat Goals Report (SHGR 2010) discussed connections between Baylands and the subtidal realm, and expressed a preference for integrated

restoration driven by the strong links between them in terms of water, sediment, organic matter, biogeochemical processes, and organisms. Most of the habitat restoration projects implemented in and around San Francisco Bay in the last 40 years have focused on single habitat types such as marshes and riparian zones. Many ecosystem processes occur at a larger scale than individual habitats, as discussed earlier in this chapter. A few large regional restoration projects have incorporated planning for multiple habitats across landscapes, including the South Bay Salt Pond Restoration Project and the Sears Point Restoration Project. Integrating restoration between subtidal and nearby marsh and upland habitats may provide ecological benefits, and the resulting interactions may result in cost savings compared to equivalent isolated restoration projects.

The most promising approach is for subtidal and intertidal restoration that results in protection of valuable Baylands. The SHGR identified several approaches to adaptation to long-term change, in particular for ways to protect marshes and other valued intertidal areas against erosion. Restoration is often expensive, uncertain, and difficult; therefore it seems logical to design restoration to capitalize on links between nearby habitats. Subtidal habitats that increase bottom friction, mainly oyster reefs and eelgrass beds, could be placed so as to attenuate wind waves and thereby buffer tidal wetlands and creek mouths from erosion. The combination of marsh restoration and nearshore subtidal habitat restoration could create local zones of sediment retention, minimizing the need for ongoing intervention. Local concentrations of oysters on constructed reefs may increase water clarity, thereby increasing the amount of light available to nearby eelgrass beds. An additional advantage to integrated restoration is to reduce the effects of habitat fragmentation. Extant marshes are small and geographically dispersed. Even after completion of the Baylands Goals Project, these habitats will not approach the extent and contiguity of pre-settlement marshes. Yet, as discussed earlier in this chapter, connectivity among habitat elements is a key feature of ecological landscapes.

Adaptation strategies will vary depending our goal: whether we are trying to increase the resiliency of existing subtidal habitat, create or expand new areas of subtidal habitats, and if we are incorporating subtidal habitat into larger efforts of nature-based adaptation of the shoreline. There are opportunities to conduct pilot projects to assess the integration of subtidal habitats into multi-objective Baylands projects to achieve multiple goals. Multi-objective and multi-habitat project designs may maximize cumulative benefits for climate change adaptation. The use of combined approaches in-lieu of a single strategy, such as sea wall construction, allows for better preparation for a highly uncertain and dynamic environment; but few projects have examined the interactive and synergistic effects and benefits of combined approaches to adaptation (Cheong et al 2012). One protection type alone can fail, and by integrating and “layering” habitat types we may create a toolbox that is more useful than just one tool alone. By thoughtfully integrating design features that can have both ecosystem service and function benefits, projects can achieve more than one outcome and can increase overall project success by including design components that support each other. The particular type of features, and their scale, dimensions, placement, and height, all must be planned with consideration of existing conditions and with future projected changes to bathymetry, substrate, tidal height, and water quality conditions. One size does not fit all with most habitat planning and restoration projects and this is especially true with constructing adaptive features.

A key component inherent in adaptation is allowing for and encouraging thoughtful and science-based experimentation in climate change restoration designs. Restoration techniques for subtidal and open bay habitats are lesser understood than for baylands, which points to the need for a thoughtful, phased approach of experimental pilot projects that can help address key science questions and generate site-specific data on restoration outcomes. Timing is critical and pushes the need to start implementing pilot, experimental integrated projects as early as possible so that we can better understand cumulative values

from linking restoration design goals and outcomes of multiple habitats in one area. Recommendations of the Intergovernmental Panel on Climate Change (IPCC), State of California Climate Adaptation Plan, and many other global and national climate change recommendations focus on the urgency of getting started as early as possible in order to pilot and test adaptation actions. In 2010 the US, Australia, and other countries committed to financially kick-starting mitigation and adaptation initiatives to produce lessons for future climate change investments and larger-scale projects (2009 Global Climate Conference).

The California Coastal Conservancy and other regional agencies are promoting project with multiple objectives of safeguarding both people and wildlife by using nature-based solutions that provide co-benefits for people, wildlife, and the economy (Coastal Conservancy 2012, Point Blue 2013). These agencies promote on-the-ground demonstration projects that implement innovative approaches or enhance understanding of effective management strategies and will potentially lead to broader change to policies, regulations, or to duplicating the effort elsewhere. Guiding principles in adaptation implementation include conserving and restoring landscape linkages and connectivity areas that will allow diverse species to move to new locations and will enhance overall species persistence (Resources Legacy Fund 2012).

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## **INTRODUCTION TO THE RECOMMENDED ACTIONS**

### **Strategies for Resilience**

Resilience management goes beyond risk management to address the complexities of large integrated systems and the uncertainty of future threats from climate change (Linkor et al 2014). Future planning should conserve and create subtidal habitats as an integral part of the aquatic transition and adjacent Bay ecosystem. Subtidal habitat is an important component of the landscape vision for the Baylands over the next century, and these habitats provide ecological benefits to the Baylands by supporting wildlife and by providing physical protection to adjacent intertidal areas. There are opportunities for thoughtful experimentation with integration of subtidal habitats into multi-objective Baylands-Open Bay pilot projects designed to achieve multiple goals. As described earlier, climate changes will influence sea level, salinity, temperature, and storm, sedimentation, and flow regimes, etc. that affect the health and function of subtidal and intertidal seabed in San Francisco Bay. This chapter has discussed the physical and biological connections between offshore shoals, the open bay waters, and the baylands; and the subtidal-intertidal transition zone in many shoreline segments provides us with an area to pilot new ideas and thoughtfully test new approaches and methods for protection of intertidal shorelines and tidal marshes. Integrated physical and biological goals can be better aligned upfront into adaptive designs that enhance and reinforce the ecosystem functions and services in the Bay Connection.

There is specific, increasing interest in designs that use the restoration of natural habitats to achieve physical benefits as well. These approaches have been coined “green infrastructure”, “soft shorelines”, “living shorelines”, etc. (see Science Foundation chapter 2. The baylands-shoreline-subtidal edge is increasingly important in planning for adaptation measures to address sea level rise and climate change. As the New York State region looks to become more resilient in the face of future storms like Hurricane Sandy, in Staten Island, planners are looking to the ecological past, which included substantial offshore oyster reefs, to define approaches for the future. A living breakwater project designed by SCAPE Landscape Architects was among the winning projects for the Department of Housing and Urban Development's Rebuild by Design contest, and the state will receive \$60 million from HUD to implement it along the South Shore coast (Orff et al 2014, [www.scapestudio.com](http://www.scapestudio.com)).

“Living Shorelines” designs can span across intertidal and subtidal areas and can take many forms, but they are generally accomplished via the strategic placement of biological habitat features (marshes, eelgrass beds, oyster beds, mudflats, etc.) and other materials (rock groins, sand beaches, shell mounds, berm islands, high tide islands, etc.) that absorb and reduce the energy of waves and currents, thereby protecting intertidal landforms (<http://www.habitat.noaa.gov/restoration/techniques/livingshorelines.html>). The features each have particular values and benefits for different species, and have the potential to provide ecosystem services such as wave attenuation and sediment accretion that may help reduce impacts to the baylands from stressors like sea level rise and increased storm frequency. The approach also can enhance valuable ecosystem functions such as nesting, feeding, and high tide refugia by adding habitat structure, increasing habitat availability of particular substrates at certain tidal heights that benefit specific species, creating linkages between habitat types, and generating substantial food resources for a variety of birds, fish, and wildlife that use the Bay Connection.

Some living shorelines projects are still based on somewhat traditional hard structure (massive amounts of granite, rock, breakwaters, etc.) that then have limited habitat components incorporated into the design or expected outcomes. Some of these efforts have been criticized for lack of ecosystem connectivity components and lack of monitoring (Pilkey et al 2012). While pilot projects in San Francisco Bay are also testing the use of artificial structures and hybrid approaches, there is a targeted approach to better define what the living shorelines concept means in this system- including careful and thoughtful use of fill, integrating multiple habitats and habitat connectivity into the design goals and outcomes, and setting co-equal goals of increasing both physical and biological benefits through careful monitoring and assessment.

Multi-objective and multi-habitat project designs may maximize cumulative benefits for climate change adaptation. By thoughtfully integrating design features that can have both habitat restoration and physical protection benefits, projects can achieve more than one outcome and can increase overall project success by including design components that support each other. For example, eelgrass beds offshore from a marsh may help provide a variety of benefits: attenuating waves (Koch 2001), providing food resources for waterfowl, providing a healthy native substrate for Pacific herring spawning (CDFW Annual herring surveys: <https://www.dfg.ca.gov/marine/herring/seasonsummaries.asp>), and functioning as a habitat corridor for fish and salmonids as they move between the open bay and the baylands ([www.sfbaylivingshorelines.org](http://www.sfbaylivingshorelines.org), Oct 2013 Monitoring Report). Oyster reefs designed as living breakwaters may have the potential to keep pace with sea level rise (Rodriguez et al 2013). Oyster beds offshore may provide these same benefits, and placement of shell reefs near to eelgrass beds may improve the success of both efforts- oyster filtering can improve turbidity which can help improve light attenuation needed for eelgrass growth (Oct 2013 LSP). Marine vegetated habitats (seagrasses, salt marshes, macroalgae, and mangroves) occupy .2% of the ocean surface but contribute to 50% of carbon burial in marine sediments. Canopies dissipate wave energy and high burial rates raise the sea floor, buffering the impacts of rising sea level and wave action that are associated with climate change (Duarte et al 2012). Like eelgrass beds, macroalgal beds provide both physical habitat and food for numerous organisms, and they can also alter flow fields, providing small organisms with shelter from currents and predators, and can trap sediment, alter sediment chemistry, and provide a substrate for spawning (Subtidal Habitat Goals Report, State Coastal Conservancy 2010). The wrack produced by eelgrass and macroalgae is an important food source for invertebrates living interstitially on beaches, mudflats, and marshes. These invertebrates in turn provide a food source for shorebirds and many other species along the shoreline and lower edges of tidal wetlands. Restoration of multiple habitat types adjacent to one another can provide benefits that may help facilitate successful restoration of each single habitat type, and achieve greater cumulative results.

Restoring adjacent habitat types with different benefits may support the various dynamics of the bay connection, and result in higher overall ecosystem functioning of the baylands and physical protection at the same time. This type of thoughtfully integrated design planning can greatly increase cost-effectiveness, by designing efficiencies into multi-objective projects that result in “low or no-regrets” restoration- ie achieving habitat restoration benefits while experimentally testing new approaches to climate change adaptation.

Because there are multiple data gaps and unknowns, it is imperative that adaptation proceed with a phased, experimental approach in order to learn what methods will be successful.

### [Link to the Subtidal Goals Recommendations – Integrated Habitats and Living Shorelines](#)

The SHGR identified several approaches to adaptation to long-term change, in particular for ways to protect marshes and other valued intertidal areas against erosion. As mentioned, one offshore/onshore adaptation technique in the pilot phases of experimentation in San Francisco Bay is the concept of living shorelines. The approaches can vary and can utilize a diverse suite of habitat restoration techniques from subtidal to upland at different elevation and tidal heights to restore, enhance, and create natural habitat for species that use the Bay Connection (including salmonids, sturgeon, sharks, herring, striped bass, rockfishes, anchovies, sardines, bottom fish, etc.).

### [Design Features will Vary According to Site Conditions](#)

One size does not fit all with most habitat planning and restoration projects, and this is especially true with constructing living shorelines as adaptive features. The effect or benefits obtained relate directly to site conditions and the type of habitat features and methods used. The particular type of features, and their scale, dimensions, placement, and height, all must be planned with consideration of existing conditions and with future projected changes to bathymetry, substrate, tidal height, and water quality conditions. For example, one meter tall oyster reefs may provide substantial wave attenuation at a shallow site but would not have any effects on wave heights if placed in deeper water at the Golden Gate. Species tolerances must be factored in as well.

Some sites with broad shallows may lend themselves to horizontal levees, eelgrass beds, oyster beds, sand beaches, etc. Steeper sites may have limited options, as depth and slope play a large role in both wave attenuation and habitat suitability for particular species. A variety of site conditions in San Francisco Bay will require a variety of approaches, with some steep, high-wave-action sites still requiring traditional hard structures like seawalls and breakwaters for protection. There is an opportunity to explore “hybrid” designs, modified approaches such as using oyster shell to create living breakwaters, or varying materials and orientations on seawalls to support algal and invertebrate habitat. These kinds of innovative techniques should be explored, in order to maximize habitat values at the bay connection and related physical conditions of the baylands and the urban edge.

### [Pilot Project Examples](#)

Current living shorelines pilot projects in San Francisco Bay are testing the use of eelgrass beds, native oyster reefs, artificial reef structures, woody debris, re-graded low marsh shorelines, and restored sand beaches and coarse gravel beaches. Preliminary data from the projects show substantial success in achieving multiple objectives- some examples include:

San Francisco Bay Living Shorelines Project: a one-acre eelgrass and oyster reef restoration project constructed in July 2012 at -1' MLLW has had more than two million native oysters settle in the first year, and an increase of more than ten new species using the site including bay shrimp, crabs, white sturgeon, steelhead, bay gobies, black oystercatchers, wading birds, and others. Up to 25% of the oysters are reproducing, along with shrimp, Dungeness crabs, nudibranchs, and bay gobies. There has been a significant increase in foraging wading birds at the site, indicating an increase in fish prey. The oyster reef structures reduce wave energy by 30-50%, depending on the water levels, with the strongest effect at mean tides ([www.sfbaylivingshorelines.org](http://www.sfbaylivingshorelines.org), Monitoring Report October 2013).

Arambaru Island Restoration Project: The re-contouring of this marsh island has reduced erosion and created haul-out areas for harbor seals and roosting areas for shorebirds and other wildlife. Placement of rock, gravel, and woody debris has helped to create roughness that slows wave action ([www.richardsonbayaudubon.org](http://www.richardsonbayaudubon.org)).

### Adaptation Recommendations

Adaptation within the Bay Connection should include methods to anticipate and respond to changes that are likely to occur over the long term, and pilot experimental demonstration projects to test ideas and gather data on results that could be scaled up. Several adaptation approaches should be tested to address specific stressors and improve the overall health of the bay connection and the Baylands.

These might include:

- Develop integrated restoration approaches that have sound experimental designs with a careful eye towards sediment dynamics, water flow, and physical and biological benefits.
- Integrated project designs that include multiple habitats and multiple objectives should be tested and can achieve cumulative benefits. Project design objectives should include both biological and physical goals when possible.
- Attention should be paid to both ecosystem services and ecosystem functions when planning specific habitat features.
- Pilot multi-objective, experimental subtidal restoration projects to assess whether these approaches can protect adjacent marshes and enhance the bay connection through increased food and habitat resources.
- Experimental projects should test a variety of designs in a variety of site conditions to learn what can work and at what scales.
- Share results broadly with resource agency staff, academic agencies, and consultants so that similar projects can get initiated at additional sites and so information can inform scaled-up efforts.
- Acquire land to act as buffer zones, e.g., around streams that provide significant sediment to the Bay or provide habitat for anadromous fish.

- Acquire or cause abandonment of islands in the Delta and Suisun Marsh that are particularly vulnerable to flooding. Some of the islands in these regions would, if flooded, cause a substantial increase in area inundated by tides, thereby increasing salinity penetration and altering tidal heights in other areas. The vulnerability of these islands introduces uncertainty about future conditions that could be reduced by selective flooding or strengthening of levees.
- Establish a program to anticipate and prepare for the consequences of the impending invasion by quagga and zebra mussels. This would go beyond existing efforts to slow the spread of these species by the California Department of Fish and Wildlife (CDFW; <http://www.dfg.ca.gov/invasives/quaggamussel/>)
- Finalize and implement the draft rapid-response plan in the CDFW's Invasive Species Management Plan (2008; <https://nrm.dfg.ca.gov/FileHandler.ashx?DocumentID=3868>, accessed 24 March 2014) for identifying and attempting to eradicate newly introduced species, and expand it to include estuarine species.

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## **SUMMARY**

The most important changes expected in the Bay Connection are those with the highest likelihood and biggest impact on the Baylands:

- Changes in the sediment budget (addressed mainly in Science Chapter 2).
- Changes in the salinity distribution arising from a variety of causes (see Science Chapter 2).
- Shifts in species composition in all habitats (due to introductions) and range shifts (due to changes in temperature and salinity), together with the ensuing (but generally unpredictable) ecological reorganization.
- Continuing increase in water clarity with increasing potential for large phytoplankton blooms and eventual eutrophication, and possibly expansion of harmful algal blooms.

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# Science Foundation Chapter 3

## Appendix 3.1 – Case Study

### Delta Smelt (*Hypomesus transpacificus*)

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#### DESCRIPTION OF THE SPECIES

The delta smelt, *Hypomesus transpacificus*, is a small, endemic, pelagic fish that lives in the Delta during spawning and early life, and in brackish waters in the Delta and Suisun Bay during its juvenile to early adult stages. Delta smelt is listed as an endangered species by the state and a threatened species by the federal government. Most of the population indices of this fish show strong declines over the last few decades.

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#### CRITERIA FOR SELECTION OF THE SPECIES

Delta smelt is the most important species in the debate about the environmental effects of water diversions and outflow requirements in the Delta. High catches of this species in the fish salvage facilities associated with the export pumps have led to reductions in pumping and consequently to water shortages. Concern over potential effects of persistently low outflow in late summer-fall has led to efforts to understand these effects and at least potentially to manage them. The Bay-Delta Conservation Plan (BDCP) is developing detailed plans for restoration of marshes in the Delta and Suisun Bay with the intent of improving conditions for delta smelt and other estuarine fishes. The likely success of these plans hinges on whether restoring marshes will help to support delta smelt.

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#### OTHER INFORMATION ABOUT THE SPECIES

The ecology of delta smelt is discussed in numerous papers, notably Bennett (2005). Delta smelt have a 1-2 year life cycle, with most fish spawning at 1 year and relatively few surviving to 2 years. The 2-year-old spawners may provide a hedge for the population against recruitment failure in a single year. Most spawning occurs in the freshwater Delta, although during wet years delta smelt may spawn in the Napa and Petaluma Rivers.

Spawning occurs in a narrow window of temperature in spring. Adhesive eggs are attached to substrate, and larvae hatch at about 4-5 mm length. Larvae begin to feed on small zooplankton such as copepod nauplius larvae, transitioning to adult copepods and later to cladocerans, mysids, and amphipods as they grow. Larvae and early juveniles move from freshwater into brackish water in June-July, and adults begin to move back to freshwater around the end of December.

Delta smelt are pelagic. They are more abundant in shallower regions than deeper channels (Aasen 1999, Bennett et al. 2002), and appear to be mainly surface-oriented. Delta smelt appear to be chronically food limited, as suggested by statistical and simulation modeling and the low abundance of zooplankton in the upper estuary (Kimmerer 2008, Maunder and Deriso 2011, Rose et al. 2013).

The summer-fall habitat of delta smelt historically has been the low-salinity zone (LSZ, a salinity range of approximately 0.5 to 5). This region is typically found in Suisun Bay to the western Delta, more landward during dry years and later in summer. The US Fish and Wildlife Service's 2008 Biological Opinion for delta smelt showed that the position of the LSZ in fall in recent wet years had been further landward than in previous wet years, as a result of a shift in export pumping from spring to fall. This resulted in a requirement to establish an adaptive management program for control of the salinity distribution in fall.

A portion of the population is present throughout the year in the Cache Slough- Liberty Island complex of the north Delta. This may be a case of alternative life histories within a population that can help a species survive in a highly variable environment (Sommer et al. 2011). The smelt are close to their thermal limits, and may be taking advantage of high turbidity and more abundant food (Sommer et al. 2011).

Abundance data for delta smelt are gathered by several long-term monitoring programs. The annual abundance index from the California Department of Fish and Wildlife's fall midwater trawl program shows abundance from 2004 to 2012, except for a pulse of high abundance in 2011, to be lower than in any year from 1967 to 2003. The moderately high abundance in 2011 coincided with late spring storms, high summer-fall freshwater flow, and more favorable conditions in several respects including higher than normal abundance of plankton. Although this rebound suggested some resilience in the population, it was short-lived and the conditions that apparently allowed for this rebound would be difficult to replicate during years with more normal seasonal hydrographs.

The putative link between delta smelt and marshes is that restored marshes may supply food for delta smelt. This is part of the reason why the Bay-Delta Conservation Plan (BDCP) calls for restoration of an area of tidal marshes that would exceed the wetted area of the Delta. The argument is that marshes are highly productive, that there is tidal exchange with the adjacent estuary, and that therefore some fraction of this productivity should be exported to subsidize the foodweb of the estuary.

The issue of connectivity between marshes and open waters is much broader, and has been analyzed far longer (Odum 1980), than can be addressed here (see main chapter on biota). In most cases primary production within marshes is likely to exceed consumption, resulting in some export of organic matter, but this organic matter may or may not be biologically available. The degree to which a marsh can provide a food subsidy to pelagic fish depends on a host of factors including the rate of tidal exchange, bathymetry, what benthic and littoral species become established, and the life histories of the particular fish.

Regardless of the particular mechanism linking delta smelt to marshes, there are two great uncertainties about this link. First, the ideas about tidal marsh restoration are more general than suggested above, and they include the idea of providing fish with a more diverse mix of habitats than the Delta in particular now provides. This idea is ecologically appealing, and if the BDCP's ambitious plans come to fruition there would certainly be more variety of habitats in the Delta and Suisun Marsh than there are now. There is little direct evidence that this will be effective for the specific purpose of supporting delta smelt. However, the use of the Cache Slough complex year-round by some delta smelt suggests that there is some value to alternative, more physically complex habitats.

The second consideration is the extent of colonization of shallow areas by unwanted organisms such as waterweeds and bivalves. Either of these could negate the benefits of a restored marsh, and both are likely in some areas.

## REVIEW OF LONG-TERM EFFECTS

Delta smelt already appear to suffer from high summer temperatures. In late summer, when the temperature gradient through the estuary is largest, the low-salinity habitat of delta smelt is furthest landward. This suggests a habitat squeeze by which the smelt cannot apparently occupy water of higher salinity but do not survive or grow well at high temperature.

Turbidity is an important habitat attribute for delta smelt. Abundance of delta smelt is weakly related to turbidity at the point of capture. Young delta smelt will not feed in clear water. The pre-spawning migration of adult delta smelt in winter appears to be related to the first flush of turbid water coming down the rivers after the first storms of the fall-winter. Although the exact mechanisms of this sensitivity to turbidity are unknown, the decreasing turbidity in the estuary, particularly the Delta, may be a factor in the ongoing decline of delta smelt.

Food supply is also a limiting factor for delta smelt. The fundamental problem of low productivity of the open-water foodweb (see plankton case study) is unlikely to be solved by restoration in the Delta. The potential increase in productivity through upgrading the wastewater treatment plants remains speculative, but investigations are underway and planned.

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## OTHER STRESSORS

Losses of delta smelt to the south Delta export facilities may cause substantial mortality (Kimmerer 2008, 2011, but see Miller 2011). These losses may be eliminated by a change in the point of diversions, although the other consequences of that change (e.g., to hydrodynamics) have yet to be analyzed. There may be some effects of contaminants, but these are probably localized and episodic.

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## FACTORS THAT MAY AFFECT SPECIES RESILIENCE

As an annual species, delta smelt is highly vulnerable to episodic events.

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## LIKELY CLIMATE CHANGE IMPACTS AND RISKS

Declining abundance due to high temperature, more landward salinity distribution in summer-fall, and continuing low food supply.

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## MANAGEMENT ACTIONS TO BE CONSIDERED

None recommended for the region included under the Update.

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## UNCERTAINTY AND KNOWLEDGE GAPS

- Planned marsh restoration, changes in wastewater treatment, and the shift in the point of diversion may have effects on delta smelt but these are unknown.

- The suitability of habitat in the Napa and Petaluma Rivers for spawning and rearing, as an alternative to the Delta during wet years, is unknown but could be important for long-term maintenance of the population.

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# Science Foundation Chapter 3

## Appendix 3.1 – Case Study

### Dungeness Crab (*Metacarcinus magister*)

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#### DESCRIPTION OF THE SPECIES

Dungeness crab, *Metacarcinus magister*, is a part-time resident of the Bay, and juveniles in particular are abundant in shallow to intertidal areas and eelgrass beds.

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#### CRITERIA FOR SELECTION OF THE SPECIES

Dungeness crabs may use marshes during development and molting, and abundance varies with ocean climate.

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#### OTHER INFORMATION ABOUT THE SPECIES

General information about the life cycle of Dungeness crabs is available from Emmett et al. (1991), and data on the abundance of age-0 and age-1 juveniles in open waters of the estuary are available from the otter trawl samples in the San Francisco Bay Study.

Dungeness crabs spawn in the coastal ocean and carry their eggs until they hatch. The planktonic larvae go through six stages before molting to the demersal juvenile stage, in which stage they move into nearshore waters and estuaries. The movement is correlated with onshore transport of near-bottom waters during upwelling (Hobbs et al. 1992).

Estuaries appear to be nursery habitats and juveniles can grow faster in estuaries than on the open coast, either because of higher summer temperature or better food resources. They use a variety of habitats including eelgrass beds. Intertidal habitat is important to young Dungeness crab in Willapa Bay, WA (Holsman et al. 2003, 2006). After a year or two in estuaries the crabs move into more saline reaches of estuaries or into the coastal ocean.

Dungeness crabs are found year-round in San Francisco Bay but they are most abundant as age-0 juveniles from May to December. Their distribution in the Bay is broad but they are most abundant at salinity > 30, and rare at salinity < 8. The abundance of adults increases with increasing depth. There is relatively little information about their use of marshes, although they are common in eelgrass beds in the Bay.

## **REVIEW OF LONG-TERM EFFECTS**

Catches in the Dungeness crab fisheries along California fluctuate cyclically and they may respond to ENSO cycles (Botsford et al. 1982). This may or may not translate to a similar response to any long-term shifts in the frequency or intensity of upwelling.

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## **OTHER STRESSORS**

Robust sport and commercial fisheries. The commercial fishery is controlled through limited entry.

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## **FACTORS THAT MAY AFFECT SPECIES RESILIENCE**

Probably high, since the crabs are opportunistic and live in a variety of habitats.

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## **LIKELY CLIMATE CHANGE IMPACTS AND RISKS**

Dungeness crab abundance is associated with low temperatures in the California Current, and therefore they are more abundant during the cool phases of climate cycles. To the extent that these cycles will change in frequency or duration, there may be changes in the recruitment of crabs to the Bay. Risks to climate effects within the Bay are low, with a possible increase in response to greater penetration of salinity.

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## **MANAGEMENT ACTIONS TO BE CONSIDERED**

Encourage continuation of San Francisco Bay Study monitoring.

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## **UNCERTAINTY AND KNOWLEDGE GAPS**

The importance of the estuary to Dungeness crab is unknown.

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# Science Foundation Chapter 3

## Appendix 3.1 – Case Study

### California Grunion (*Leuresthes tenuis*)

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#### DESCRIPTION OF THE SPECIES

California grunion *Leuresthes tenuis* is found only along the outer coast of California and Baja California. It is a relatively small atherinid fish that spawns at high tide on sandy beaches; the eggs incubate out of water until the next semilunar high tide. The main habitat range is between Ensenada, Mexico, and Pt. Conception, California.

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#### CRITERIA FOR SELECTION OF THE SPECIES

San Francisco Bay is the type locality for this endemic species (Ayres, 1860), although none was seen there again until 2001 (Skinner, 1962; Jahn, 2004). From 2001 to 2008, California grunion was present throughout the year in San Francisco Bay and spawned on multiple beaches. Grunion formed a substantial part of the diet of Least Terns nesting at the Alameda colony during this time (Elliott et al., 2007).

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#### OTHER INFORMATION ABOUT THE SPECIES

California grunion abundance was monitored by the CA Department of Fish and Wildlife's San Francisco Bay study and found throughout the Central and South areas of the bay between 2001-2007. Dropped fish were identified by PRBO Conservation Science at the Alameda Least Tern Colony. Beaches that were used as spawning sites included Chrissy Field Beach in the Golden Gate National Recreation Area, Crown Memorial State Beach in Alameda, Albany Bulb, Oakland Middle Harbor, Emeryville, Foster City, Seaplane Lagoon at the Alameda Naval Station, and Roberts Landing in San Leandro. Beaches were monitored for spawning runs by Grunion Greeters from 2005 on (Martin et al., 2007). Spawning runs occurred from May to August during semilunar high tides.

No California grunion were observed on the outer coast around San Francisco Bay. It appears that San Francisco Bay was colonized from southern California in 2001 and that individuals remained within the bay throughout their lives (Johnson et al., 2009).

Within San Francisco Bay, mature California grunion were smaller in length and mass than those from southern California, and egg diameters and clutch volumes were significantly smaller. Few in this population appeared to survive more than one year and spawning season, however farther south they may survive two to four years.

California grunion feed on zooplankton throughout life (Horn et al., 2006).

## REVIEW OF CLIMATE CHANGE EFFECTS ON SPECIES

During ENSO and other regional climate cycles, one would expect a return of California grunion to San Francisco Bay during warmer phases. Even if the outer coastal waters remained cooler, this species can complete its life cycle within the bay.

It is likely that northern bays are thermal refuges for California grunion (Johnson et al., 2009). In 2005, California grunion colonized Tomales Bay, a northern range extension (Roberts et al., 2007). This population was observed spawning on the beach each year through 2009, but not since. California grunion has also appeared in Monterey Bay sporadically through the past 60 years (Phillips, 1943; Spratt, 1981; Yoklavich et al., 2002).

Rising sea levels will cause changes in coastal ecosystems including sandy beaches, the critical spawning habitat of this species. Armored shorelines increase the threat of habitat loss for sandy beaches. Areas where managed retreat is possible will be crucial for maintaining potential spawning habitat in the future. Within San Francisco Bay, the beaches where California grunion spawned are public shorelines, managed for recreation. The sandy substrates were artificially placed and are variable in grain size but all these beaches are relatively narrow at high tide.

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## OTHER STRESSORS

California grunion tolerates salinity fluctuations in brackish water above approximately 16 ppt but does not survive in extremely low or high salinities (Reynolds et al., 1976; Matsumoto and Martin, 2008), as could occur with changes to freshwater flows into San Francisco Bay or increased evaporation on shore.

California grunion is prey for many species of marine teleosts, elasmobranchs, and shorebirds (Gregory, 2001).

California grunion is sensitive to many different anthropogenic chemicals including oil spills, and the larvae have been used in bio-assessments of toxicity (Valentine and Soulé, 1973; Winkler et al., 1983; Hose and Puffer, 1984; Newton et al., 1985; Borthwick et al., 1985; McCoy, 1998). If waters outside the bay are too cold for this species, grunion may be unable to escape unhealthy levels of chemicals even if they are only present temporarily.

Sandy beaches are necessary for spawning habitat. The eggs incubate buried in sand, completely out of water, and remain vulnerable to human activities while developing.

California grunion is subject to a recreational fishery (Walker, 1952). The closed season, April through May, was set for southern California populations and does not protect the primary spawning period for grunion in San Francisco Bay. Smaller runs and smaller clutch sizes indicate that recreational fishing should be closed throughout the spawning season for this species within San Francisco Bay.

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## FACTORS THAT MAY AFFECT SPECIES RESILIENCE

Resilience is low at the northern limits of the habitat range at this time, but may improve with warming.

## LIKELY CLIMATE CHANGE IMPACTS AND RISKS

The most likely climate effects include sea surface temperatures inducing northern shifts in habitats, changes in salinity on spawning beaches from increased rainfall, and loss of critical habitat with sea level rise on sandy beaches, particularly in areas with large amounts of armoring.

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## MANAGEMENT ACTIONS TO BE CONSIDERED

Full closure of the recreational fishery is recommended, along with management practices that maintain the upper beach above the mean high tide, avoidance of mechanized maintenance or vehicles over spawning sites (Martin et al., 2006).

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## UNCERTAINTY AND KNOWLEDGE GAPS

It is likely California grunion will return to San Francisco Bay, but uncertain when that will occur. The effects of sea level rise on the width and slope of the man-made beaches within the bay are not known. Increased sand transport into the bay may enhance beach resilience.

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# Science Foundation Chapter 3

## Appendix 3.1 – Case Study

### Pacific Herring (*Clupea pallasii*)

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#### DESCRIPTION OF THE SPECIES

Pacific herring, *Clupea pallasii*, is the only fish that supports a commercial fishery in the Bay. Pacific herring occur in the coastal ocean for most of the life cycle but spawn, and larvae rear, in the Bay. They are captured for their roe, and eggs are harvested directly using kelp as a spawning substrate.

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#### CRITERIA FOR SELECTION OF THE SPECIES

The spawning areas of Pacific herring are vulnerable to aspects of long-term change, especially sea level rise, changing salinity distribution, and possibly temperature. Schools of adult herring are likely affected by conditions in the coastal ocean.

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#### OTHER INFORMATION ABOUT THE SPECIES

Pacific herring occur throughout the north Pacific Ocean in stocks associated with estuaries used for spawning. Schools of adult herring enter the Bay in winter and spawn by depositing sticky eggs on hard substrates and vegetation in the low intertidal and shallow subtidal areas of the Bay. Eggs are vulnerable to predation by fish, birds, and mammals, and can be harmed by excessive sediment loads. Early survival is related to salinity, with the highest survival at low to moderate salinity and low survival at oceanic salinity (Griffiths et al. 2004). Young herring rear within the Bay through the larval and early juvenile stages. Recruitment of young herring to the adult population is controlled mainly by survival from spawning through the larval stage (O'Farrell and Larsen 2005).

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#### REVIEW OF LONG-TERM EFFECTS

Probably the most important long-term effects on Pacific herring will be due to changes in salinity. Since eggs and early larvae survive best at salinity well below oceanic levels, and spawning requires substrates that are most available in Central Bay, a trend toward a more landward position of the salinity field would result in lower recruitment. A possible exception to this would occur if eelgrass were to form large beds further up-estuary than where they now occur, providing lower-salinity spawning habitat. Increases in salinity during the winter spawning season are likely to arise through changes in the tidal prism, higher sea level, and changes in water project operations to capture more of the winter runoff than is now removed.

Changes in the coastal ocean would affect herring and in particular the proportion of the population that spawns in the Bay. These changes are not well constrained by climate forecasts, nor is the response of herring abundance to changes in the ocean well understood.

Sea level rise will alter the nearshore bathymetry, which would have either positive or negative effects on spawning habitat for herring. Higher temperature may reduce survival of eggs spawned in the intertidal zone.

A reduction in suspended sediment may improve survival of herring eggs. Winter suspended sediment concentrations depend more on flood-borne sediments than the decreasing pool of sediments in the Bay, but sediment loading through the Delta has decreased (Wright and Schoellhamer 2004).

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## **OTHER STRESSORS**

Eggs spawned on pier pilings containing creosote do not survive well (Vines et al. 2000). The eggs are vulnerable to damage by oil spills.

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## **FACTORS THAT MAY AFFECT SPECIES RESILIENCE**

Probably high. The herring fishery is closely monitored and well regulated.

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## **LIKELY CLIMATE CHANGE IMPACTS AND RISKS**

Changes in the salinity distribution during winter could have a large impact on early survival and subsequent recruitment.

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## **MANAGEMENT ACTIONS TO BE CONSIDERED**

If the anticipated landward shift in salinity were to occur, recruitment might suffer as a result of the poor overlap between suitable spawning substrate and suitable salinity. This may warrant action to provide more spawning habitat, perhaps in connection with construction to adapt to rising sea level and development of living shorelines.

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# Science Foundation Chapter 3

## Appendix 3.1 – Case Study

### Longfin Smelt (*Spirinchus thaleichthys*)

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#### DESCRIPTION OF THE SPECIES

The longfin smelt is an anadromous Osmerid native to the San Francisco Estuary (Moyle 2002), which is home to its southern-most reproducing population (Rosenfield and Baxter 2007). Historically, the longfin smelt was one of the most abundant pelagic species, supporting a commercial “white bait” fishery (Skinner 1962). Since the 1960s, the abundance of longfin smelt in the San Francisco Estuary has declined consistently and precipitously except during brief periods of higher outflow (Thompson et al. 2010). In the early 2000s, the abundance of longfin smelt, and 3 other upper estuary pelagic fishes, took another sharp decline that has yet to be fully explained (Thompson et al. 2010, Mac Nally et al. 2010). This simultaneous multi-species decline has come to be known as the Pelagic Organism Decline (Sommer et al. 2007). The longfin smelt is classified as ‘threatened’ under the California Endangered Species Act (OAL 2010) and the San Francisco Estuary component of the west coast population was assessed as warranting federal threatened status, but its listing was precluded due to higher priority work (USFWS 2012).

Within the San Francisco Estuary, increased outflow (or reduced X2 position) during its spawning and early rearing periods (i.e., December through June) continues to produce a significant increase in longfin smelt abundance (Sommer et al. 2007, Kimmerer et al. 2009, Baxter et al. 2010), but more recent analyses suggest that increased flows during spring (March-May) produce the strongest effect (Mac Nally et al. 2010, Thompson et al. 2010, Nobriga and Rosenfield unpublished). Longfin smelt abundance is also significantly influenced by adult spawner number (Baxter et al. 2010, Nobriga and Rosenfield unpublished). This outflow-abundance relationship remains the strongest and most consistent of those observed for organisms within the estuary (Kimmerer et al. 2009). Future water management in support of this relationship will likely benefit longfin smelt and a suite of other estuarine and anadromous fishes, as well as some invertebrates (see Kimmerer 2002, CDFG 2010).

In tributaries to the San Francisco Estuary, longfin smelt probably seek sand or gravel substrates not far above brackish water (CDFG 2009a&b) and spawn, incubate and hatch during the coldest period of the year (Baxter 1999, CDFG 2009a&b). Using the presence of newly hatched larvae and projecting back in time, assuming a 25-day incubation duration, CDFG (2009a) estimated that longfin smelt spawn from November through April and that spawning is initiated about when water temperatures drop below 16°C and becomes widespread at water temperatures < 13°C. After a long incubation (ca. 25-40 days; CDFG 2009a), longfin smelt hatch into buoyant larvae that are dispersed by tides and outflow such that the distribution of late stage larvae and young juveniles centers around 2 psu salinity (Dege and Brown 2004, Hobbs et al. 2010, Kimmerer et al. 2009). With growth and development, longfin smelt disperse broadly throughout the Estuary, including to marine waters (Baxter 1999, Moyle 2002), but avoiding regions where water temperature exceeds 22°C (CDFG 2009a). In recent years this movement by young of the year longfin smelt (though both age classes show shift) to more marine rearing habitat appears to be the

dominant life history strategy as exemplified by both a higher salinity distribution (Figure 10, Contreras et al. 2011) and proportionally many more fish being caught in central San Francisco Bay than more upstream embayments (Baxter et al. 2010). It's unclear whether the observed distributions represent an actual shift, possibly in response to poor upper Estuary feeding conditions as argued for northern anchovy (see Kimmerer 2006) or result of better survival in marine regions. Earlier migration to marine waters could be one means of avoiding increasing summer temperatures in the upper Estuary.

Considering the use of marsh habitats, Brown (2003) described longfin smelt as a transient species found almost exclusively during the cooler winter and spring periods of the year. He described longfin smelt as migrating through the Suisun Marsh to spawn upstream. However, both Brown (2003) and CDFG (2009b) present information indicating that freshwater sources, within marshes, that flow substantially during winter and spring can attract adults during their spawning migrations; presence of adults and, later in the year, larvae strongly indicated that spawning took place in Cordelia Slough or its tributary Green Valley Creek within western Suisun Marsh (CDFG 2009b); similarly, Meng and Matern (2001) found longfin smelt larvae most abundant in Cordelia Slough, and their abundance associated with low temperatures and high outflows. Thus, some marshes may be seasonally important to longfin smelt.

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## LIKELY CLIMATE CHANGE IMPACTS AND RISKS

As global air temperatures increased in the past several decades, so have surface water temperatures (see summary in Cloern et al. 2011). Scaled to the San Francisco Estuary, model projections indicate variable but significant future increases in Delta water temperatures, sea level and salinity, and concurrent significant declines in precipitation and runoff, both only in the more severe temperature increase scenario (Cloern et al. 2011). These scenarios suggest four primary modes of effect on longfin smelt: 1) increasing water temperatures reduce the length of the spawning period; 2) increasing winter and spring water temperatures potentially create a metabolic demand among larvae prior to food availability; 3) increased summer and fall water temperatures reduces habitat quality of the upper Estuary in general and marshes in particular; and under the more extreme temperature change scenario, 4) reductions in runoff, particularly in spring, causing a negative effect on recruitment.

Warming winter water temperatures in the range of 1.5 to 4°C, depending upon model assumptions, could reduce the duration of the longfin smelt spawning period by 2-3 weeks, as inferred from modeling results depicting earlier spring warming (cf., delta smelt mean spawning window date; Brown et al. 2013) and an assumption that fall cooling would not be similarly affected (e.g., no increase in stress temperatures, Brown et al. 2013). It's not clear whether this reduction in the longfin smelt spawning window (longfin smelt likely spawn in water temperatures <16°C, CDFG 2009a) would have a population level effect -- longfin smelt spawning and early rearing would remain synchronous with earlier outflows projected by models-- but it would affect the temporal range available for reproductive hedge-betting.

Metabolic demand in fishes is in part a function of water temperature (Adams and Breck 1990). Increased winter and spring temperatures can be inferred from model projections of 2 -3+ week earlier mean dates of the delta smelt spawning window (Brown et al. 2013). Such temperatures will increase the metabolic demand of longfin smelt larvae, but it's unclear whether future patterns of lower trophic bloom timing will support this higher demand. Even small changes in early survival are likely to have profound affects on abundance.

Negative effects on summer and fall longfin smelt habitat quality in the upper Estuary can be inferred from Cloern et al. (2011) and Brown et al. (2013), both of whom projected substantial increases in the number of high mortality days (>25°C) within the Delta under all future scenarios. Although longfin smelt tolerate higher salinity and are found farther down in the Estuary, it is likely that these high temperatures will propagate downstream (see Shellenbarger and Schoellhamer 2011) and negatively affect longfin smelt habitat. Unlike delta smelt, longfin smelt juveniles and sub-adults have the option to move to cooler marine waters (Baxter 1999). This temperature effect on habitat suitability will likely be more pronounced in shallow, marsh waters where volume is low, heat exchange is high, and water exchange is limited.

Finally, predicted changes in runoff -- shifting from spring to winter, increased variability and a potential for decline under certain scenarios (Cloern et al. 2011) -- does not bode well for longfin smelt whose reproductive success is strongly tied to high winter and spring outflows (see Kimmerer et al. 2009 and references therein). A shift in runoff from spring to winter will likely continue to associate high flow periods with the more restricted longfin smelt reproductive window, possibly maintaining the current beneficial associations. Increased outflow variability, given the species current extremely low abundance, places additional stress on the already stressed population, particularly in the case of a protracted drought. In the absence of a drought, if the future includes a general decline in runoff, it's also likely to include an increased human demand (Cloern et al. 2011), resulting in greater pressure to allocate the reduced flow to uses outside the Estuary, producing negative effects on the longfin smelt population.

These conclusions should be viewed as inferentially based individual- or population-level responses derived from examining modeled subsets of possible future environmental outcomes and using best professional judgment of the author; they should not be construed as predictions. Our understanding of the temperature tolerances of longfin smelt life stages is extremely limited, and we know nothing of their metabolic demands, so inferences in these areas are highly speculative.

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# Science Foundation Chapter 3

## Appendix 3.1 – Case Study

### Northern Anchovy (*Engraulis mordax*)

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#### DESCRIPTION OF THE SPECIES

Northern anchovy (*Engraulis mordax*) is the most abundant fish in San Francisco Bay. It is a relatively small, planktivorous schooling fish with large populations along the Pacific coast.

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#### CRITERIA FOR SELECTION OF THE SPECIES

Northern anchovy is the most abundant fish in the estuary, especially in the more saline reaches of the estuary, principally San Pablo to South Bay. Coastal ocean populations vary in response to regional ocean climate, so long-term shifts are likely. Because they are so abundant, anchovies are found in all regions of the estuary west of the Delta, including marsh channels, where they may be major consumers of plankton.

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#### OTHER INFORMATION ABOUT THE SPECIES

Anchovy abundance has been monitored throughout the estuary since 1980 by the CA Department of Fish and Wildlife's San Francisco Bay study. This program provides an annual index based on monthly samples taken at about 50 stations throughout the estuary.

Anchovies are highly abundant in the coastal ocean and provide an important food source for larger fish as well as birds and marine mammals. Abundance fluctuates with ocean climate, with higher levels during years of strong upwelling compared to El Niño years (Chavez et al. 2003). There are several subpopulations along the Pacific coast, including a northern (Oregon to central California) and central (central to southern California) subpopulation (Parrish et al. 1985). These subpopulations are divided at about the latitude of San Francisco Bay, but the Bay population is linked genetically to the central California subpopulation (Vrooman et al. 1981).

The California Cooperative Fisheries Investigations (CalCOFI) organization has conducted an egg survey for anchovy in the southern subpopulation. Data for anchovies in the ocean near San Francisco Bay are indirect, including landings in the fishery and bycatch from a rockfish survey (Kimmerer 2006). The Bay population index does not fluctuate as much as, and is uncorrelated with, indices of the biomass of the coastal population (Kimmerer 2006). Thus, the Bay population may be separate from, or subject to other factors than the coastal population. There may nevertheless be a quantitative connection between these populations, but we are unable to detect it since there has not been a monitoring program adequate to characterize anchovy abundance in coastal waters near the Bay.

Anchovies are abundant year-round in the estuary. The Bay population appears to have been stable over the last three decades, but the fraction of the population in low-salinity waters declined abruptly in summer

1987 when plankton populations dropped following the spread of the introduced clam *Potamocorbula amurensis* (Kimmerer 2006). This was probably due to movement by the fish rather than mortality.

Anchovies feed mainly as visual predators on zooplankton but may switch to filter-feeding and consume both zooplankton and large phytoplankton such as diatoms. Thus, fluctuations in anchovy abundance, seasonality, or distribution are likely to have substantial effects on estuarine foodwebs. Specifically, rising or falling populations in the coastal ocean off the Bay mouth could affect Bay populations and thereby lower trophic levels in the Bay.

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## REVIEW OF LONG-TERM EFFECTS

Coastal populations fluctuate in response to ENSO and other regional climate cycles, with higher abundance during the cool phases. The likely response to climate change is difficult to predict since changes in ENSO, PDO, and NPGO and associated changes in wind, ocean temperature, and upwelling are poorly constrained by current climate models. Any trend toward reduced upwelling would likely result in a decrease in anchovy abundance; however, such a decrease may be difficult to detect in the coastal ocean because of high variability and a lack of monitoring. However, changes in the Bay population would be readily detectable by the existing monitoring program.

Any effect on food supply (plankton) would probably affect anchovy strongly, as would major changes in the abundance of predators. See the phytoplankton case study for long-term influences at the base of the planktonic foodweb. Striped bass, a major predator on anchovy, may be influenced by climate through their tendency to migrate to sea when the coastal ocean is warm. This may have caused the mid-1970s decline in striped bass population size in the Bay, which previously had been attributed to export pumping (Kimmerer et al. 2001). A decline in striped bass might allow for an increase in anchovy, although it is only one of many predators and the others are likely to respond differently.

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## OTHER STRESSORS

Availability of food is important to any fish, and the evidence cited above indicates that anchovies are sensitive to food supply. Contaminants probably have little effect on these fish since they feed low on the food chain and would not be likely to concentrate contaminants as do striped bass and other high-level predators.

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## FACTORS THAT MAY AFFECT SPECIES RESILIENCE

Unknown

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## LIKELY CLIMATE CHANGE IMPACTS AND RISKS

The most likely climate effects would occur through changes in the food supply. There are no significant risks to the long-term maintenance of anchovy but, as key elements in the estuarine foodweb, changes in their abundance would have substantial effects on both predators and prey in the open waters of the estuary and in marshes

## MANAGEMENT ACTIONS TO BE CONSIDERED

Ensure continuity of the Bay Study monitoring program. Support some monitoring effort in the coastal ocean.

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## UNCERTAINTY AND KNOWLEDGE GAPS

- Trajectory of the ocean population and the strength of its link to the Bay population. The former uncertainty is caused by both environmental variability and lack of monitoring data; the latter uncertainty could be reduced if monitoring in the ocean were initiated, or through a research program.
  - Responses to changes in foodweb productivity are more predictable at the level of anchovies themselves, but not at the level of the foodweb.
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# Science Foundation Chapter 3

## Appendix 3.1 – Case Study

### Oyster Beds and Reefs

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#### DESCRIPTION OF THE SPECIES

This case study focuses on the native Olympia oyster, *Ostrea lurida*. Much of the information herein was obtained from an Appendix to the Subtidal Habitat Goals report (Zabin et al. 2010).

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#### CRITERIA FOR SELECTION OF THE SPECIES

Oyster beds can provide a variety of ecosystem services, although their current low abundance means they probably do not play a huge ecological role in today's estuary. Nevertheless, there is potential for restoration, particularly in living shorelines together with eelgrass for adaptation to higher sea level, protecting shorelines without increasing wave reflection. Oyster beds may be vulnerable to several aspects of long-term change in ways that could interact with marshes.

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#### OTHER INFORMATION ABOUT THE SPECIES

These oysters are found mainly on hard substrates in the intertidal zone, although their abundance in the subtidal is poorly known. Artificial reefs have been installed at several pilot restoration sites, and oysters have recruited to the reefs in large numbers. Oysters occur throughout the more saline parts of the estuary although they are generally most abundant between the San Rafael and San Mateo bridges.

Native oysters do not build tall reefs but cement and expand existing hard structures and increase roughness. Thus they can be considered "ecosystem engineers" that create or alter habitat for themselves and other species.

The functions of oyster beds comprise those due to the physical structure and those due to the living oysters. Reefs that have been created for restoration typically have a taller profile than oyster beds that have grown on existing hard substrate such as rock and riprap. By altering bathymetry, these reefs can absorb energy from waves and tidal currents, trap sediments, and alter local mixing. The structures also increase physical heterogeneity, providing habitat for a wide variety of invertebrates and fish, particularly if the structures are physically complex. They can provide spawning substrate for herring.

The living oysters provide a food resource for other estuarine organisms. They filter the water and can locally reduce phytoplankton biomass. However, given the limited extent of their habitat they are unlikely to filter as much phytoplankton as the soft-bottom clams do, except in the immediate vicinity of dense oyster reefs with limited water circulation.

The principal link to marshes seems to be the reduction in wave energy and bottom stress that a constructed reef or living shoreline would provide, protecting an adjacent marsh from erosion. It is unclear

whether the oysters themselves would play a major role in long-term maintenance and expansion of these constructed reefs. If they do, the outcome would require less maintenance than would be the case for largely abiotic reefs, which would need periodic cleaning and replacing degraded reef units.

The importance of oysters in the bay before European settlement is uncertain. However, here we are considering only future conditions and what role oyster reefs might play. The historical information would be of limited value for informing future actions given all the changes in the estuary.

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## **REVIEW OF LONG-TERM EFFECTS**

Oyster reefs may suffer negative effects from ocean acidification. Oyster shells are made of calcium carbonate which dissolves at low pH. The larval stages of oysters are particularly vulnerable to acidification (Hettinger et al. 2012). The key uncertainty is whether those effects override or are additive to the naturally high variation in pH in the estuary.

Changes in the distribution of salinity can affect oysters and if upstream hydrologic changes (e.g., reduced snowpack) produce more frequent floods the frequency of die-backs may be higher. Rising temperature may cause desiccation of oysters, particularly newly settled spat, high in the intertidal zone but not lower. Oyster fecundity is temperature dependent, and warmer springs might result in earlier spawning and larval release, but the population-level effects of earlier reproduction are not clear. Disease incidence in the Bay is currently low but incidence or its impacts may increase if oysters are already stressed. Reduced sediment supply might benefit oyster populations through reduced sedimentation on the reefs; however, sediment supply to particular locations may remain high because of local erosion and transport.

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## **OTHER STRESSORS**

Oyster drills can cause considerable mortality to native oysters, particularly small ones. Oyster drills, which is non-native in SF Bay, are found in high abundance in the South Bay and in Richardson Bay. Competition or overgrowth by fouling species (many of which are also non-native) may reduce recruitment and slow growth. Low levels of dissolved oxygen (DO) also appear to depress growth and increase mortality; although present levels of DO are high, this could change if the estuary becomes eutrophic in the future.

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## **FACTORS THAT MAY AFFECT SPECIES RESILIENCE**

Oyster populations rebounded quickly at a San Rafael restoration site after the 2006 floods. However, the overall resilience of the oyster populations can't be known without more information about their population structure. Although the larval period is long enough that oyster larvae should be broadly mixed within the estuary before settlement, there is some evidence of population structure. Although settlement of oysters onto newly implanted substrates has been rapid in restoration sites, ~40% of the suitable substrate surveyed by the UC Davis group is unoccupied, implying either a shortage of larvae or other processes inhibiting settlement.

## **LIKELY CLIMATE CHANGE IMPACTS AND RISKS**

Enumerated above.

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## **MANAGEMENT ACTIONS TO BE CONSIDERED**

The use of oyster reefs in living shorelines should continue to be investigated as part of an adaptation strategy for rising sea level and reduced sediment supply. Because of the lack of knowledge about oyster population dynamics, oyster restoration actions should be undertaken in an experimental framework so that the effort is not wasted and the knowledge necessary for continuing restoration can be accumulated.

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## **UNCERTAINTY AND KNOWLEDGE GAPS**

- The roles of the oysters themselves in the structure, function, and long-term resilience of artificial reefs.
  - The function of the reefs in providing habitat and reducing shoreline erosion, and how this varies with location.
  - Abundance of oysters in the subtidal
  - Connectivity among oyster populations, including genetic variability as well as ecological connectivity (i.e. coherence in dynamics across the metapopulation).
  - Population dynamics of oysters.
  - The role of oyster reefs in providing substrate for introduced species.
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# Science Foundation Chapter 3

## Appendix 3.1 – Case Study

### Plankton

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#### DESCRIPTION OF THE GROUP

This case study considers phytoplankton and zooplankton, emphasizing productivity of foodweb support to upper trophic levels. Phytoplankton or planktonic algae provide most of the energy supply to the pelagic foodweb of the estuary and therefore also to the benthos and some marsh organisms. Much of the phytoplankton production is consumed by microzooplankton and clams. Larger zooplankton consume phytoplankton and microzooplankton.

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#### CRITERIA FOR SELECTION OF THE GROUP

Planktonic production is key to the overall productivity of the estuary and to the food available to fish. In Suisun Bay and the western Delta, the food supply to fish is a severe constraint and has probably contributed to declines in fish and other species. Long-term trends in phytoplankton and zooplankton reveal substantial shifts in abundance and species composition, and further changes can reasonably be anticipated in the future. In the past four decades phytoplankton production has varied considerably in response to various influences including climate in the coastal ocean, introduced bivalves, increasing water clarity, and nutrients. Zooplankton abundance has responded most strongly to alien species introductions.

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#### OTHER INFORMATION ABOUT THE GROUP

Plankton in general either drift passively or swim at low speeds relative to the movement of the water (note: they are not all weak swimmers, but their small size limits their swimming speed). The strong tidal currents throughout much of the estuary mix the plankton between channels and shoals, and vertically except when the channels are stratified by salinity or when some zooplankton undergo vertical migration.

The species composition of plankton changes with salinity from a freshwater assemblage to an oceanic assemblage, with a few species present only at intermediate salinities. Phytoplankton include single cells with sizes from 1 to >100  $\mu\text{m}$ , and chains and aggregates that can be visible to the naked eye.

Phytoplankton is a heterogeneous collection across a very wide range of taxonomic groups including some cyanobacteria ("blue-green algae"). Long-term monitoring has shown the dominance of certain species in certain regions, and how that dominance has shifted over time ({Cloern, 2005 #4748; Cloern, 2012 #7097}).

The zooplankton includes microzooplankton (smaller than 0.2 mm) and the larger mesozooplankton (0.2 – 2 mm) and macrozooplankton (>~2mm). Microzooplankton includes single-celled organisms such as

ciliates and flagellates, and larval and juvenile stages of copepods. Mesozooplankton includes juvenile and adult copepods, cladocerans, and larvae of various organisms such as clams, crabs, barnacles, and fish.

### Phytoplankton Production

Plant production (or productivity) is the rate at which organic matter is produced by photosynthesis within a given area or volume. Phytoplankton production is the largest source of available organic carbon to the open waters of the estuary (Sobczak et al. 2002). Production by other aquatic plants such as attached algae and vascular plants (see Submerged Aquatic Vegetation (SAV) case history) is probably much smaller than that by phytoplankton because of the limited extent of SAV and the high specific growth rates of phytoplankton. Production by benthic microalgae has not been studied much in the estuary, and may also be important in shallow areas (Cornwell et al. 2014).

For the purposes of this discussion, we consider net production, i.e., photosynthesis less respiration. Net production is the product of biomass, the quantity of phytoplankton (usually as organic carbon) per unit estuarine volume or surface area, and specific growth rate, i.e., the daily growth expressed as a fraction of the biomass. These two factors can be considered separately in how they are measured and in what environmental factors affect them.

Several long-term monitoring programs routinely measure chlorophyll concentration, a proxy for biomass. Production has been measured in shorter-term studies, mostly using uptake of radioactively labeled carbon (e.g., Cole and Cloern 1987, Kimmerer et al. 2012). Production is not routinely monitored but can be estimated from biomass together with incident light and water clarity, which are monitored.

Phytoplankton production is influenced by a wide variety of factors, most of which are likely to change in the coming decades. Specific growth rate depends on the following factors:

Light level: Phytoplankton growth rate increases with light level up to its maximum rate, which depends on the other factors below. In this currently turbid estuary, growth rate is typically under severe light limitation in the deep channels except during periods of salinity stratification (Cloern 1984).

Phytoplankton growth rate can be high over shoals, and in the surface layer when channels are stratified.

Water depth: In shallow water light may penetrate to the bottom, allowing phytoplankton to grow at all depths. In deep water, the depth-averaged growth rate of phytoplankton can drop below the respiration rate such that net production is negative. The only way this can be supported in the long run is by mixing between regions of high (shoals) and low or negative (channels) production, although periods of stratification in channels can trap phytoplankton in the surface waters and cause positive growth (Cloern 1984, Lucas et al. 1999).

Nutrient concentrations and composition: In many estuaries high nutrient concentrations cause eutrophication, by which excessive growth of phytoplankton produces excessive organic matter, resulting in shading out of SAV, and in some cases depletion of oxygen in bottom waters. In the San Francisco Estuary over the last several decades eutrophication has not been a problem despite high nutrient concentrations, mainly because of high turbidity.

This situation is no cause for complacency for three reasons. First, the waters of the estuary are becoming clearer (Kimmerer 2004) as the pool of available sediment has declined (Schoellhamer 2011). At some point this could allow for phytoplankton growth to accelerate to the point where large blooms occur and eutrophication is possible.

Second, there is some evidence that the form of nitrogenous nutrients may affect the growth rates of some phytoplankton. Specifically, nitrate uptake is inhibited by moderate to high ammonium concentrations ({Dortch, 1990 #6468}), and some diatoms may grow faster when using nitrate as a nutrient source than when using ammonium ({Dugdale, 2007 #5650}). The exact relationship of ambient phytoplankton growth to ammonium concentration is uncertain, and the putative link between ammonium concentration and suppression of diatom blooms is controversial. However, ammonium concentration is usually high enough to limit nitrate uptake because of releases by wastewater treatment plants, and the quantity released has increased in parallel with changes in phytoplankton biomass and species composition (as well as numerous species introductions and other changes).

Third, there is also evidence that harmful algal blooms (HABs) may be increasing in the estuary, possibly as a result of increasing water clarity, changing nutrient concentrations, and changing water residence time (Lehman et al. 2005).

Species composition: Phytoplankton growth rates and their responses to environment (light, nutrients, temperature, salinity) are species-specific. For example, some species of diatom can grow very rapidly under conditions of high nitrate, low to moderate temperature, moderate light level, and moderate turbulence. Some cyanobacteria can fix nitrogen from atmospheric N<sub>2</sub> gas, and can therefore grow when nitrogenous nutrients are scarce. Motile species such as dinoflagellates, or buoyant forms such as the toxic cyanobacteria *Microcystis*, a HAB species, can move into favorable depths for light or nutrients. All of these differences make it very difficult to predict which species of phytoplankton will become abundant in blooms under current conditions or as a result of long-term change.

There has been a long-term trend in San Pablo Bay to the Delta toward smaller and possibly less nutritious phytoplankton species. This trend has been linked alternatively to size-selective grazing by the introduced clam *Potamocorbula amurensis* and changes in nutrient composition. Smaller cells are less available for grazing by zooplankton, with the result that foodweb efficiency has decreased. This, together with the overall low productivity of this part of the estuary, implies poor foodweb support for fish (Kimmerer et al. 2012, Cloern and Jassby 2012).

The expansion of *Microcystis* in the Delta roughly coincided with the pelagic organism decline in the upper estuary, indicating a possible link through the foodweb. Ongoing field-based experiments on the effects of *Microcystis* on zooplankton that are key foodweb organisms in the Delta will help to determine the effect of these blooms, although laboratory experiments suggest a strong negative effect on survival (Ger et al. 2010).

Temperature: Physiological rates of phytoplankton and other ectothermic organisms depend on temperature, generally increasing with temperature to a species-specific maximum and then declining sharply. Photosynthetic rate responds more strongly to ambient variation in light than variation in temperature, while respiration rate is strongly sensitive to temperature. There is no information on the temperature response specifically for phytoplankton in the San Francisco estuary.

Salinity: Phytoplankton move with the water and therefore do not experience changes in salinity the way that marsh or benthic organisms do. A given species is most abundant over a range of salinity and its abundance declines at higher and lower salinity. This pattern is due to some combination of salinity tolerance of individual species, which can affect growth, and factors that affect biomass (below).

Biomass accumulation or depletion depends on specific growth rate, grazing, and hydrodynamic transport to and from other regions of the estuary.

Grazing by zooplankton and bivalves removes most of the phytoplankton production in the estuary. In the northern estuary, and previously in South Bay, grazing by bivalves controls the accumulation of

biomass (Cloern 1982, Lucas et al. 1999, Thompson 2005, Kimmerer and Thompson 2014). Grazing by the clam *Potamocorbula amurensis* has been high since its introduction in 1986, and phytoplankton biomass in the northern estuary has been high only during a few brief spring blooms (Kimmerer and Thompson 2014). Grazing by this clam has also depressed abundance of some copepod species and probably microzooplankton (Kimmerer et al. 1994, Greene et al. 2011).

Grazing by microzooplankton is also an important loss term for phytoplankton, and at times can exceed that by bivalves, while grazing by mesozooplankton such as copepods is the third most important grazing term (Kimmerer and Thompson 2014). In the South Bay grazing by clams declined sharply around 1999, resulting in an increase in phytoplankton biomass. The decline in clam abundance was apparently due to an influx of benthic-feeding fish which in turn resulted from a shift in ocean climate (Cloern et al. 2007).

The distribution of benthic grazers shifts rather slowly with seasonal movement of the salinity gradient, while the plankton move with the water and therefore most of the individuals in a population are not directly affected by salinity. However, an indirect salinity effect can arise when the temporal patterns of freshwater and tidal flow place a plankton population over a shoal or a channel, and in or out of contact with a clam bed.

Transport occurs through advection, i.e., the gradual seaward movement of the water due to river input, and dispersion or horizontal mixing, which moves salt, sediment, and plankton from areas of high abundance toward areas of low abundance. Mixing is largely effected by the tides and influenced by the salinity distribution (see section on salinity in Chapter 2 and on sediment movement in Chapter 4).

Transport can bring freshwater or oceanic phytoplankton into the estuary, mix phytoplankton between favorable and unfavorable salinity or depth regimes, and spread a buildup of biomass from shallow areas to deep areas where blooms are unlikely (Lucas et al. 1999). It can also bring phytoplankton into or out of proximity with benthic grazers, outfalls, water intakes, shoals, and other geographically static habitat elements that may affect biomass.

### Zooplankton Production

Little is known about the long-term patterns of abundance or taxonomic composition of microzooplankton because, despite their importance in the estuarine foodweb, they have never been monitored. Microzooplankton consume bacteria, phytoplankton, and other microzooplankton. They are the second most important grazers in the northern estuary after clams (Kimmerer and Thompson 2014), and the most important food for some mesozooplankton (Rollwagen Bollens and Pen 2003, Bouley and Kimmerer 2006, Gifford et al. 2007).

As with phytoplankton, production of zooplankton is the product of biomass and growth rate. Under conditions of plentiful food, growth rate increases with increasing temperature below the upper thermal limit for each species, and also varies with species. Growth of microzooplankton presumably responds to temperature as well as phytoplankton or bacterial biomass, but previous experiments in the low-salinity zone showed relatively little response which was interpreted to mean that food is usually adequate for microzooplankton (York et al. 2011), and their biomass may be limited by consumption by clams (Greene et al. 2011). However, the growth rates of copepods including their nauplius larvae are chronically food limited (Kimmerer et al. 2005, W. Kimmerer, unpub.). This can be interpreted to mean that the availability of food for small fish is limited by the productivity of phytoplankton, at least in the upper estuary.

Micro- and mesozooplankton are also subject to transport by tidal and net (river-derived) currents but the larger organisms (> ~0.5 mm) are capable of tidal and daily migrations that effect retention within some region of the estuary (Kimmerer et al. 2002, Kimmerer et al. in prep.). Microzooplankton are also removed

through grazing by clams, and this can be a significant loss to the populations despite strong escape responses of the larger organisms (Kimmerer et al. 1994, Greene et al. 2011).

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## REVIEW OF LONG-TERM EFFECTS

Many of the controlling factors discussed above are likely to change in the long run with changing climate and human interventions. The long-term changes likely to have the greatest effect on phytoplankton are:

Changing water clarity due to shifts in the estuary-wide sediment budget, with potential contributions from changing wind speed, tidal prism, and water depth. Although the main concerns with increasing clarity are for eutrophication and harmful algal blooms, Suisun Bay and the western Delta are currently in a state of low productivity, and a trend toward eventual eutrophication would have to pass through a period of moderate to high production before production becomes excessive. Thus, for that region, which seems most affected by the increase in water clarity, the concern over eutrophication may be premature.

However, the trajectory of phytoplankton production depends on the continued high impact of grazing by introduced clams, and if that grazing were to be reduced (as it was in South San Francisco Bay), the trophic status of the estuary would shift toward eutrophication.

Changing nutrient inputs with increases in human population and improved treatment. This effect is likely to be strong locally, but the broader-scale implications of a significant reduction in total nutrient inputs, or in ammonium inputs specifically, are uncertain. Equally uncertain is the likely response of blooms of harmful algae such as *Microcystis*. Some of these species do well in high-nutrient conditions, but the level of nutrient concentrations that would inhibit these blooms is unknown.

Changes in the coastal ocean: Plankton move between the coastal ocean and the Bay, affecting species composition and thereby responses to environment within the Bay. Any changes in the coastal ocean through, e.g., changes in upwelling intensity, local current patterns, or temperature will likely affect the species composition and condition of plankton entering the Bay. However, the likely magnitude and direction of such changes are poorly understood.

Species introductions, particularly of bivalves or other filter-feeders or their predators, or of some species of zooplankton. These shifts could occur naturally through range expansions and shifts in ocean climate, or through human activities. For example, the anticipated changes in the salinity distribution (section 2.F) will lead to shifts in species distributions with consequences that would be difficult to predict.

Particular concerns for species invasions to the estuary are quagga and zebra mussels, which are likely to arrive in the estuary soon. Although these are considered freshwater species, they can tolerate some salinity. Furthermore, if established they would consume the freshwater phytoplankton that now provide a subsidy to the brackish parts of the estuary where grazing by *Potamocorbula* has suppressed phytoplankton biomass, leading to further declines in the brackish foodweb. Previous studies of zooplankton in ballast water, a likely vector for introducing planktonic organisms, did not reveal species that would be likely to have a major impact in the estuary (Choi et al. 2005), although that does not rule out future high-impact invasions through ballast-water discharge.

Changing water circulation due to a reduction in freshwater flow into the estuary during summer may make parts of the Delta less physically dynamic. This would make the Delta more suitable for blooms of

*Microcystis*, which could affect foodwebs in Suisun Bay, and also of the waterweed *Egeria densa*, which may remove nutrients from the water.

Temperature effects at the ecosystem scale are difficult to predict because the responses are species-specific. Some harmful algal species (possibly *Microcystis*) may be more suited to high temperature than low. Most of the introduced zooplankton species are most abundant in summer, suggesting tolerance to high temperature. The reproductive season of these zooplankton and some benthic grazers will likely be extended, extending the time during which net phytoplankton growth is negative.

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## **OTHER STRESSORS**

Contaminants such as herbicides may affect phytoplankton in the estuary. One study showed this effect to be episodic and patchy during 1997 in the Delta, where large quantities of herbicides are applied (Edmunds et al. 1999). Effects of a variety of contaminants on zooplankton have been reported, but these also appear to be sporadic (Werner et al. 2000).

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## **FACTORS THAT MAY AFFECT SPECIES RESILIENCE**

Individual species may or may not be resilient to changes in their immediate environment. The assemblages along the salinity gradient comprise a variety of species with overlapping salinity and seasonal distributions. This diversity of forms and responses to environment mean that foodweb support by plankton should be resilient to most kinds of change. However, the assemblage present in the northern estuary in 1986 appeared resilient, but by 1993 it had been completely altered by clam grazing and introductions of several copepods, and possibly also by a changing nutrient regime.

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## **LIKELY CLIMATE CHANGE IMPACTS AND RISKS**

The most significant impacts are likely to come not from climate but from species introductions and increasing water clarity. The risks cannot be calculated.

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## **MANAGEMENT ACTIONS TO BE CONSIDERED**

Anticipate and plan for the likely impacts of quagga and zebra mussels in the Delta. Maintain the Interagency Ecological Program's monitoring for lower trophic levels and water quality variables in the upper estuary. Expand or supplement that monitoring program to include South and Central Bay and expand coverage of San Pablo Bay. Add a microzooplankton component.

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## **UNCERTAINTY AND KNOWLEDGE GAPS**

- Thresholds for harmful algal blooms and effects on zooplankton.
- Likely effects of quagga and zebra mussels.
- Distribution, abundance, and trajectory of microzooplankton throughout the estuary.

- Distribution, abundance, and trajectory of mesozooplankton in the lower estuary.

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# Science Foundation Chapter 3

## Appendix 3.1 – Case Study

### Rocky Intertidal Organisms

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#### DESCRIPTION OF THE GROUP

Rocky intertidal habitat in the Bay is largely confined to Central Bay, particularly around the Golden Gate, Angel and Alcatraz Islands, and in parts of South Bay and Carquinez Strait. This habitat hosts a suite of organisms including attached algae and animals such as sponges, bryozoans, tunicates, hydrozoans, anemones, barnacles, mussels, and oysters. Numerous other invertebrate animals (e.g., crabs) and fishes (e.g., prickly sculpin) forage in rocky intertidal areas at high tide.

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#### CRITERIA FOR SELECTION OF THE GROUP

Rocky intertidal areas are exposed at low tide and therefore to high air temperature, so warming is likely to affect these areas. Species introductions have also had a great influence on rocky intertidal assemblages.

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#### OTHER INFORMATION ABOUT THE GROUP

Rocky intertidal habitat with its associated organisms is characteristic of the Pacific coast from Baja California to Alaska. Within central California there is a characteristic suite of species that can colonize rocky intertidal areas, and the actual species found at a given location depends on wave energy, sun exposure, elevation, steepness of the shore, and various biotic interactions. The species assemblage within San Francisco Bay is likely similar to those in similar environments on the outer coast, although these habitats have not been surveyed.

A total of 162 species of attached algae have been reported from surveys within the estuary, most attached to intertidal hard substrate including naturally occurring rock and artificial substrate. Most of these were species also found on the open coast, while 33 were classified as estuarine and five were accidentally introduced (Silva 1979, Josselyn and West 1985).

Native (*Mytilus trossulus*) and invasive (*M. galloprovincialis*) mussels occur on rocky intertidal substrates in the Bay, where they apparently compete for space (Schneider and Helmuth 2007). The introduced species seems better able to withstand high temperature, particularly during exposure at low tide, and is more abundant in exposed locations (Schneider and Helmuth 2007). This emphasizes that it is the interaction among species that will determine the outcomes of climate change, rather than only the physiological responses and stress tolerance of each species.

## **REVIEW OF LONG-TERM EFFECTS**

High air temperature can cause desiccation, potentially lowering the upper elevation that can be occupied by less-tolerant species. This is likely to result in rearrangements of species assemblages due to competitive and predatory interactions. In particular, thermally sensitive species will be forced lower in the intertidal where they will be subject to heavier predation by subtidal and low-intertidal organisms such as starfish (Helmuth 2002). High temperature and low oxygen content in isolated pools may result in loss of biota there.

Rocky intertidal assemblages can be expected to maintain their vertical zonation in relation to the range of tides, and therefore to keep up with rising sea level if there is accommodation space available.

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## **OTHER STRESSORS**

Oil spills can cause severe damage to rocky intertidal areas.

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## **FACTORS THAT MAY AFFECT SPECIES RESILIENCE**

Probably high.

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## **LIKELY CLIMATE CHANGE IMPACTS AND RISKS**

High temperature – both air and water

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## **MANAGEMENT ACTIONS TO BE CONSIDERED**

None recommended

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## **UNCERTAINTY AND KNOWLEDGE GAPS**

Little is known about rocky intertidal habitats in the Bay apart from the two algal surveys. A survey of these habitats may be warranted.

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# Science Foundation Chapter 3

## Appendix 3.1 – Case Study

### Salmon

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#### DESCRIPTION OF THE SPECIES

This case study covers Chinook salmon (*Oncorhynchus tshawytscha*) and sea-run steelhead (*Oncorhynchus mykiss*). Under natural conditions these salmon spawn in gravel-bedded rivers in the watershed and migrate through the estuary on their way to sea, where they rear for one to several years before returning to their natal streams to spawn. Most Chinook and many steelhead are hatchery-reared fish, which are released in the rivers or in the estuary to begin their migration. A small proportion of these fish spawn in some years in local tributaries to the Bay, such as the Napa River.

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#### CRITERIA FOR SELECTION OF THE SPECIES

Salmon are strongly affected by several aspects of climate. Their spawning and rearing habitats are affected by changes in freshwater flow and temperature. Survival and growth in the ocean depends on ocean conditions, and the warm water and low productivity of El Niños can cause poor survival. Survival of young salmon through the Delta can be very low, and may be affected by various threats including predators, contaminants, and high temperature. High export flows to the south Delta pumping facilities, and unfavorable circulation patterns, can reduce survival of migrating salmon (Newman 2003, Kimmerer 2008). Salmon that spawn in the smaller tributaries are more vulnerable to variability in local flow and temperature, but may provide a source of recolonization and genetic variability to the riverine populations.

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#### OTHER INFORMATION ABOUT THE SPECIES

A monograph by Williams (2006) provides most of the information summarized here. Most of the available information is on Chinook salmon; less is available for steelhead because of their lower abundance and diffuse and variable migratory patterns.

Life cycles of salmon are complex, adaptable, and variable. Chinook salmon spawn in gravel-bedded rivers of the Central Valley, either tributaries in the Sierra foothills or in the mainstem Sacramento River below Keswick Dam. Eggs deposited in gravel hatch and the fry remain within the gravel before emerging when they begin to feed. Fry may remain in their natal reach or drift downstream and rear in other habitats. Most fry go through the process of smolting during their first 6 months – 1 year, by which they become ready to encounter salt water, and begin a rather leisurely movement toward the sea. The fish rear in the ocean for 1-5 years before returning to the rivers where they spawn and die. Life cycles of steelhead are somewhat more complex, in that they migrate to sea at a variety of ages or not at all, and can return to sea after spawning. Steelhead generally overwinter at least once, and are therefore vulnerable to low flow and high temperature during summers. Steelhead are also genetically linked to resident rainbow trout

populations that spawn in the same rivers, indicating some plasticity between the migratory and resident life histories.

Four runs or races of Chinook salmon have been identified based on the timing of adult migration upstream: winter-run, spring-run, fall-run and late-fall run. These runs are genetically distinct and have distinct patterns of timing of upstream and downstream migration, and spawning and rearing habitat. Winter-run Chinook are listed as endangered and spring Chinook as threatened under both state and federal legislation, and Central Valley steelhead are listed as threatened under federal legislation.

Hatcheries, built mainly to replace natural production lost to damming of rivers, produce a large number of fish particularly in the fall-run stock, which is the mainstay of the ocean fishery. The fall-run Chinook population is largely a hatchery stock, although many of these fish spawn naturally (Barnett-Johnson 2007, Johnson et al. 2012). Molecular or isotopic methods are necessary to identify individual fish to run once they have migrated out of the river, so results of studies before 2000 are generally ambiguous as to run.

Young Chinook salmon are present in the estuary throughout the year, but are most abundant in the Delta during April-June. Although estuarine rearing is common in other stocks of Chinook salmon (Healey 1991), estuarine rearing seems less important in Central Valley Chinook. Chinook smolts entering brackish water take about 3 weeks to traverse the estuary, and generally grow less in weight than expected for their length, indicating a shortage of food within the estuary (MacFarlane 2010). Salmon generally consume insects and various crustaceans such as copepods and amphipods, switching to fish as they grow. The low weight of the fish upon entering the ocean is consistent with low productivity in the estuary.

Chinook salmon use marshes extensively in other Pacific estuaries, but in the San Francisco Estuary they appear to be most abundant in shallow waters of the Delta (Grimaldo et al. 2009). A sampling program in the China Camp marsh turned up no salmon in over 13,000 fish (Visintainer et al. 2006), while six Chinook salmon were collected out of 9452 fish caught in marshes of the Napa and Petaluma Rivers and the western Delta (Gewant and Bollens 2012). Chinook salmon are abundant during spring in samples collected by the San Francisco Bay Study and comprise on average about 4% of the catch throughout the year. They are about equally abundant in shoal and channel stations, with no trend in catch with depth of the sampling station.

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## **REVIEW OF LONG-TERM EFFECTS**

Chinook salmon in the Central Valley are at the southern end of their range and vulnerable to high temperatures exacerbated by loss of habitat due to dams. Winter Chinook once spawned in the high-elevation reaches of the Sacramento River and tributaries, where eggs and fry could develop during summer in dependably cool water. Their spawning is now confined to an artificial cold-water reach below Keswick Dam. This makes them highly vulnerable to a few years of low flow resulting in high temperature because of low reservoir level and high air temperature. A conservation hatchery program has been implemented to hedge against a catastrophic drought. Adult spring Chinook hold over summer in deep, spring-fed pools in streams draining the Lassen watershed, where they are somewhat protected from high-temperature events as long as the spring-water remains cool.

Estuarine conditions will likely continue to impair growth of Chinook salmon. However, high-flow conditions appear less favorable than low-flow conditions of strong salinity penetration into the estuary (MacFarlane 2010). This suggests that the anticipated change toward lower spring-summer outflow and greater salinity penetration may benefit salmon once they get past the Delta, but low-flow conditions within the Delta are probably deleterious to salmon, particularly with the current water export system (Newman 2003, Kimmerer 2008). The proposed change in the point of diversion of freshwater from the southern

Delta to the Sacramento River may benefit salmon by reducing entrainment in the southern Delta, but that depends on details of the design, construction, and operation of the new facilities, which have not been fully developed.

Ocean conditions have a strong influence on early growth and survival of Chinook salmon (MacFarlane 2010, Wells et al. 2012). Upwelling is associated with high abundance of krill, good survival and condition of the fish, and strong subsequent recruitment to the fishery.

Salmon populations in local streams are likely to be vulnerable to high temperature and reduced flow during the protracted dry season, possibly exacerbated by water withdrawal in the local watershed. This is particularly so for steelhead because they over-summer in the streams.

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## **OTHER STRESSORS**

Although hatchery production was instigated to mitigate for loss of habitat, hatcheries have had a devastating impact on the integrity of naturally-spawning stocks (Williams 2006, Johnson et al. 2012). Ocean harvest of the hatchery-supported fall Chinook has probably reduced the two listed stocks because the fish cannot be distinguished, although they are somewhat separated and the catches of listed stocks are probably a small fraction of their total abundance.

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## **FACTORS THAT MAY AFFECT SPECIES RESILIENCE**

Fall Chinook is maintained by hatchery production and cannot be considered resilient, although the hatcheries should be able to continue maintaining this stock. The two listed runs are highly vulnerable and have few alternatives if spawning areas become unavailable because of drought, heat, or catastrophe. Steelhead by themselves are vulnerable to the same problems; however, if their genetic link with resident rainbow trout indicates strong mixing between these life histories, the resident trout may enhance resilience of the steelhead.

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## **LIKELY CLIMATE CHANGE IMPACTS AND RISKS**

High temperature, coupled with changing water project operations and seasonal patterns of runoff, may devastate the listed Chinook runs.

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## **UNCERTAINTY AND KNOWLEDGE GAPS**

- Importance of shoals and marsh channels for salmon.
- Contribution of populations in local streams to overall populations.

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# Science Foundation Chapter 3

## Appendix 3.1 – Case Study

### Submerged Aquatic Vegetation

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#### DESCRIPTION OF THE GROUP

Submerged aquatic vegetation (SAV) includes, going from salt to freshwater: eelgrass *Zostera marina*, several species of pondweed (*Stuckenia* spp.), and the Brazilian waterweed *Egeria densa* as well as several other freshwater invasive species. The freshwater species are largely outside the domain of this report.

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#### CRITERIA FOR SELECTION OF THE GROUP

SAV generally is regarded as an important component of estuarine ecosystems and an indicator of ecological condition. SAV is generally in short supply in the lower estuary and a nuisance in the Delta (e.g., invasive species such as *Egeria densa*). Eelgrass in particular is vulnerable to several aspects of climate change, particularly rising sea level. In addition, eelgrass restoration, alone or as part of a living shoreline, may be useful in protecting shorelines and marshes.

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#### OTHER INFORMATION ABOUT THE GROUP

Most of this discussion concerns eelgrass beds, because eelgrass is the predominant SAV in the lower estuary, has the greatest potential both for climate effects and for adaptation to rising sea level, and is better known than the other SAV forms. Much of this information comes from Boyer and Wyllie-Echeverria (2010) and the Subtidal Habitat Goals Report (2010).

Eelgrass beds form in shallow areas of sandy-mud bottom, where they trap and stabilize sediments, alter hydrodynamics, and alter sediment chemistry, all of which can enhance conditions for the maintenance of the beds. They also add structure to otherwise bare areas, providing habitat for many species that may use the eelgrass for protection or feed on the eelgrass or associated biota.

Eelgrass beds occur in the saline areas of the estuary, covering about 1500 ha or 1% of the area of San Pablo, Central, and South Bays. The largest bed is just north of Point San Pablo in San Pablo Bay, but most of the other major beds are in Central Bay (Merkel and Associates 2009). These beds occur mostly within 2m of mean lower low water, presumably because of light limitation (Zimmerman et al. 1991). Most of the beds are perennial, reproducing vegetatively and also through annual seed production resulting in new shoots, except for one bed near Alameda that primarily reproduces annually by seeds. Reproduction from seeds appears to be inhibited by high winter flows accompanied by high turbidity, as occurred in 2006 (K. Boyer, SFSU, pers. comm.).

Despite their close proximity, major beds within the Bay are genetically differentiated, possibly as a result of local selection (Ort et al. 2012). This may imply adaptation to local conditions, and the authors recommended care in selecting sources of plants to be used in restoration.

A current research project (K. Boyer, SFSU, PI) is investigating the salinity tolerance and other characteristics of pondweeds, *Stuckenia* spp.. Both *S. pectinata* and *S. filiformis* are present, and they may hybridize. These species have only recently been documented in the open waters of Suisun Bay and the west Delta, although *S. pectinata* has long occurred in ponds behind levees (Jepson 1905). New beds and patches have established in the open Suisun Bay in the last five years according to aerial imagery, suggesting that conditions are becoming more favorable. Although the reason for this is unknown, salinity penetration in summer-fall has increased, apparently because of changed operations of the large water projects in the Central Valley (USFWS 2008) and light penetration may be increasing due to a now-limited supply of erodible sediment (Kimmerer 2004, Schoellhamer 2011). There are now about 400 ha of *Stuckenia* beds in Suisun Bay and the western Delta, and in some places the density of the beds is high enough to be a nuisance for boating. The plants senesce each winter and grow back from belowground tissues in spring.

The Brazilian waterweed *Egeria densa* forms large beds throughout much of the freshwater Delta. In places it chokes waterways, reduces current speeds, and traps sediment, possibly making the water clearer. Its distribution is largely outside of the scope of this report, but it probably has had a substantial impact on the Delta pelagic foodweb and therefore that of Suisun Bay. In addition, *Stuckenia* spp. overlap in distribution with *Egeria* in the western Delta and competition between these species may influence responses to changes in salinity, temperature, or water clarity (Borgnis 2013).

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## REVIEW OF LONG-TERM EFFECTS

Eelgrass beds are most vulnerable to rising sea level, which would reduce light levels at the beds and possibly also increase current speeds over the beds, reducing the efficiency with which beds capture sediments.

If suspended sediment concentrations continue to decline, eelgrass beds may be less able to trap sediments, reducing their capacity for self-maintenance and for maintenance of a sediment pool that might protect marshes and shorelines. This effect would interact with that of rising sea level, inhibiting both upward growth of the beds to maintain a constant depth through accretion and lateral growth up-slope. Lateral growth will also be inhibited in places with steep shorelines, especially where levees and seawalls reflect waves.

Water clarity is probably a limiting factor for the maximum depth of eelgrass beds, and reduced local sediment supply will allow water clarity to increase, possibly offsetting the other effects. However, the limit on depth may occur during winter-spring, when floods bring pulses of highly turbid water at the same time when new shoots are beginning to grow. High turbidity and possibly reduced salinity may explain a recruitment failure of eelgrass in San Pablo Bay during winter-spring of 2006.

A landward shift of the salinity gradient in summer may allow eelgrass beds to expand up the estuary. The other SAV species may already be responding to a landward shift in salinity in summer-fall, which is likely to increase (see section 2.1F).

Rising temperature is less likely than other changes to affect eelgrass, except possibly in some beds in shallow areas protected from waves and currents. To the extent that dissolved CO<sub>2</sub> increases relative to background, this may favor the growth of eelgrass and other SAV.

## **OTHER STRESSORS**

Oil spills can cause severe damage to eelgrass beds. The effect of contaminants on SAV is unknown, except that the Department of Boating and Waterways uses herbicides to reduce SAV buildups near boat harbors and these herbicides may have effects beyond the areas of application. Experimentally elevated nutrients do not appear to affect eelgrass (G. Santos, SFSU thesis in progress).

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## **FACTORS THAT MAY AFFECT SPECIES RESILIENCE**

Probably high; eelgrass plants can grow clonally or through seed dispersal, and substantial belowground biomass can allow for recovery from periods of unfavorable conditions such as low salinity.

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## **LIKELY CLIMATE CHANGE IMPACTS AND RISKS**

Increased sea level and reduced sediment availability are likely to have negative effects, possibly offset by increasing water clarity.

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## **MANAGEMENT ACTIONS TO BE CONSIDERED**

Continue actions to protect eelgrass beds. continue investigations into the functions of eelgrass beds, particularly their ability to trap sediments, reduce waves and currents, and protect shorelines. Continue pilot restoration projects at an intermediate scale using a scientific approach to learn about methods for restoration as well as the functions of the beds.

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## **UNCERTAINTY AND KNOWLEDGE GAPS**

The uncertainties for SAV are summarized here from those enumerated in the Subtidal Habitat Goals Report (2010):

- Specific ecosystem functions supported by eelgrass beds
- Extent of functions contributed by plants vs. physical structure
- Scaling of functions to area of restored eelgrass beds
- Response of eelgrass beds to local biotic and abiotic environment
- Limits on establishment of new beds, naturally or as restoration.
- Effect of tidal and wind-driven circulation, wakes, and suspended sediment on persistence of beds and on dispersal of seeds and recruitment.
- Ecological and genetic connectivity among beds
- Effects of bed size, fragmentation, and density of plants in an eelgrass bed on persistence and expansion
- Extent of exogenous mortality of eelgrass and controls on die-back and recovery.

- The most effective methods for restoration and maintenance of eelgrass beds with minimal intervention
- Interactions between oyster and eelgrass beds
- Distribution, abundance, and limiting factors for other SAV species

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# Science Foundation Chapter 4

## Connections to the Watersheds: The Estuarine-Terrestrial Transition Zone

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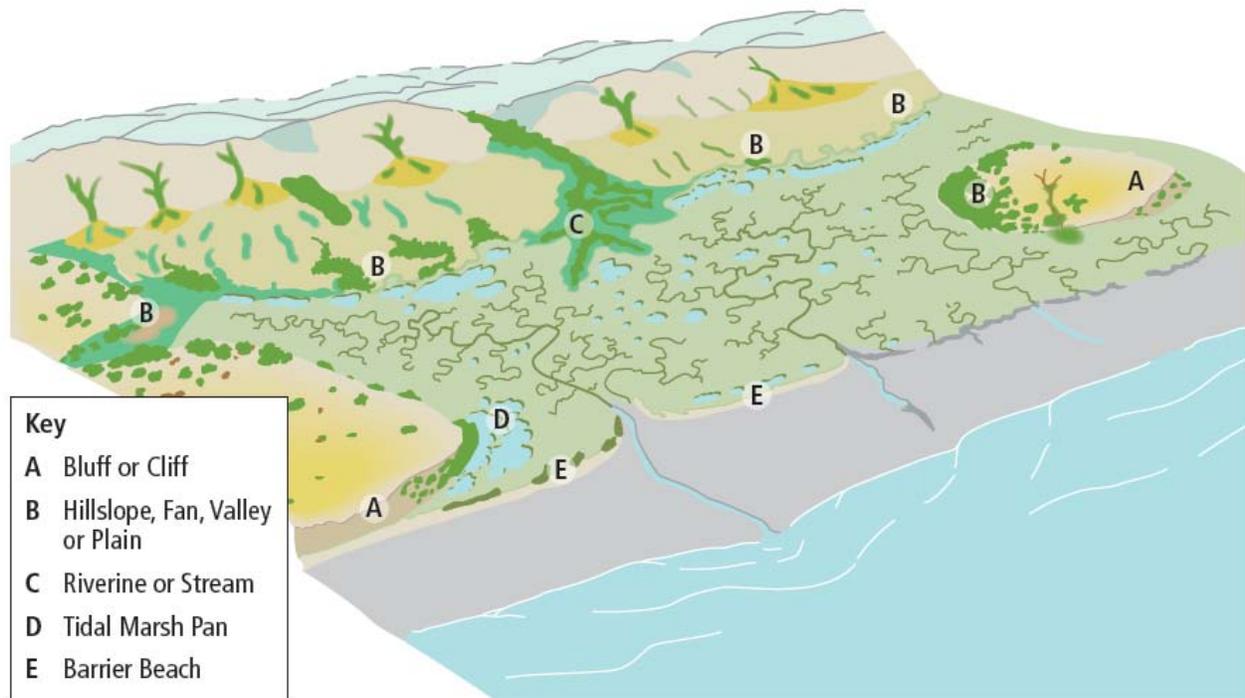
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## INTRODUCTION

Life in the Bay Area is concentrated along the bayshore. Most people live or work within half a mile of the Bay (US Census 2010). The history of people in the region living along the shoreline is evidenced by thousands of years of bayshore habitation and industry, with ancient shellmounds still apparent in shoreline parks to skyscrapers built over gold rush wharves in San Francisco. Maritime commerce connects the region to the rest of the world via seaports and airports built on the Bay edge. The remaining marshes of the Bay support most of the region's rare or endangered plants and animals (Harvey et al 1992). The edge of the Bay is packed with ecological, economic and cultural values.

The values of the bayshore are always changing. Historical values supported diking and filling the marshlands and shallows of the Bay for agriculture and commercial salt production, some of which was later converted to suburbs and other intensive land uses. This is one of the most urbanized estuaries in the world. In the urbanized areas, almost nothing is left of the natural shoreline. It has been fitted with major infrastructure for communications and power transmission, and for moving people, commercial goods, water, fuel, and wastes. This infrastructure rings the Bay, crossing through our current and former tidal marshes, crossing over most of our rivers and streams, and restricting connections between the Bay and its local watersheds. Much of the wildlife, water, and sediment from surrounding hills and valleys now move along unnatural channels through built environments to reach the Bay.



This figure illustrates a typical arrangement of the different transition zone types in a virtual landscape that represents the range in natural physiography around San Francisco Bay downstream of the Delta. The tidal salinity regime could be brackish or saline. This figure idealizes and integrates landscape characteristics of what are now Oakland, San Lorenzo, Hayward, Coyote Hills, Turk Island, and eastern Marin County. The foreground hillslopes (oak woodland top; coastal scrub on stabilized scarp behind barrier beach) are over-representing a local feature by perspective. Alluvial fans and plains (grassland) are realistically prominent. Tidal marsh pans (aka marsh ponds), freshwater non-tidal ponds (depressional wetlands), and wet meadows (slope wetlands) along streams and the baselines of hillsides are exaggerated to be visible. The tidal marsh is not part of the T-zone except for the backmarsh including the pans nearest the uplands and the barrier beach bordering the tidal flats. Natural salt ponds are larger than tidal marsh pans and are not represented in this figure. Artificial levees are also not included because they are not natural features.

The Bay's been rising too slowly to significantly constrain modern land use along its shoreline. We've only had to strengthen some levees and raise them a little higher, add pumps to drain lowlands behind the levees after floods, or raise diked lands above the Bay's current reach to protect our uses of the shoreline from gradually rising Bay waters. Public policies about the Bay have focused more on maintaining the bayshore as it is now than on its inevitable change due to sea level rise. For decades we've carefully minimized every activity that makes the Bay smaller, and now we need to figure out how to let the Bay get larger.

Climate change is a game changer. A rapidly rising Bay will change our economy, ecology, and culture (BCDC 2011, Heberger et al. 2012, Ayyub and Kearney 2012). Governmental agencies and other organizations responsible for the health of the Bay are exploring many ways to accommodate its expected

rapid rise and expansion inland and upstream. Some of the major concerns are about managing flood hazards and protecting the essential ecosystem services<sup>1</sup> of the tidal flats and marshes.

The many services provided by the Bay will be lost if the tidal flats and marshes become tightly squeezed between the rising Bay and the built environment.

The challenge is not due to sea level rise per se, but the accelerated rate of rise expected by mid-century. Coastal people have been living with a gradual rate of sea level rise for thousands of years. However, the rate of its rise is expected to accelerate (IPCC 2014). How long the Bay will rise at any predicted rate is unknown. However, prudence and common sense demand that we plan our response now.

Efforts to address the threats imposed by rapid sea level rise have begun to focus on the shore of the Bay and the area of transition between the bay and the adjoining uplands. There is a growing awareness that this estuarine-terrestrial transition zone between the Bay and the uplands, hereafter termed the “T-zone”, is needed to mitigate these threats. The T-zone is an integral part of a complete tidal marsh ecosystem (Science Foundation Chapter 2). It can provide space for the Bay to rise and expand without creating unacceptable flood hazards and without completely losing the ecological services of the tidal flats and marshes. There is also a growing recognition that the T-zone provides critical support for wildlife throughout the region, while also supporting its own unique plant and animal communities.

The interest in the T-zone has intensified since the first Baylands Habitat Goals Report in 1999. The forecasts of rising Bay waters were much less certain at that time than they are today. While there was a general appreciation of the need to restore and conserve the T-zone in the 1999 Goals Report, the broad range of T-zone services were much less understood, and our need for the T-zone to mitigate the threats of a rising Bay did not seem urgent. This update of the Baylands Goals Report provides an

### **We Aren't the First Bay Area People Facing a Rising Bay**

People have been living along the central coast of North America for at least the last 11,000 years (Erlandson et al. 2008). During this time, the ocean has risen across the river plain that is now the Gulf of the Farallones, entered the Golden Gate, and formed the Bay (Atwater et al. 1977). Until about 3,000 years ago, sea level was rising so fast that every generation of people living along the coast or Bay was probably forced to retreat inland (Bickel 1978).

As the rate of sea level rise slowed, large marshes formed around the Bay and people established permanent villages near the marshes (Nelson 1907). They harvested shellfish to eat and used the shells to construct monumental mounds. Some of the mounds were more than 3 stories tall and some were built miles inland from the shore. No one knows all the reasons for these shellmounds. Were they made in memory of ancestors forced to retreat from the rising ocean and Bay? Are there shellmounds beneath the Bay and under the Gulf of the Farallones?

The Bay is rising rapidly again. Will those of us living and working along its shore retreat inland to safety? We have more infrastructure and land use to protect than ever before. Will we build walls and install pumps to keep the Bay out? Will we build islands to live on? How will future generations regard our response to a rapidly rising Bay? What will be our legacy as the next people facing this major environmental challenge?

Climate change is creating opportunities to restore the natural functions and services of the Bayshore. They were traded away for modern land uses that are threatened by the rising Bay. We can protect ourselves from the rising Bay in ways that sustain these services into the future. Will we?

<sup>1</sup> Ecosystem services are the processes of ecosystems and their material and energy outputs that benefit people. Essential services provide such benefits as food and water, building materials and natural fuels, flood control and disease control, recreation and spiritual healing, pollution filtration, nutrient cycling, and biological diversity.

important opportunity to address more fully the need to restore and protect the T-zone now and into the future.

### Objectives

The objectives of this chapter are to define and describe the T-zone for the San Francisco Estuary downstream of the Delta, and to provide science-based guidance for conserving the ecological services of the T-zone in the context of climate change and especially sea level rise.

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## DEFINITION OF TRANSITION ZONE

The recommended definition is intended to be scientifically sound, comprehensive and practicable. It was developed from a set of guiding principles and evaluation criteria developed by the T-zone workgroup (see Appendix 4.1). This definition incorporates abundant local professional experience with t-zone evaluation and management, and it reflects the current scientific literature regarding the nature of environmental transition zones in general. The definition should be revised as necessary to reflect future advances in the scientific understanding of the T-zone and its services to people.

*The estuarine-terrestrial transition zone, or T-zone, is the area of existing and predicted future interactions among tidal and terrestrial or fluvial processes that result in mosaics of habitat types, assemblages of plant and animal species, and sets of ecosystem services that are distinct from those of adjoining estuarine, riverine, or terrestrial ecosystems.*

The T-zone as defined here does not include all the baylands. It also does not include all of the tidal marshlands. It only includes the portion of the marshlands wherein the plant community is directly and measurably influenced by terrestrial runoff and other freshwater discharges. It includes diked baylands that serve to store terrestrial flood waters, since flood control is a T-zone serviced, but it does not necessarily include other diked baylands.

The T-zone has often been visualized as the area of transition between tidal marsh vegetation and terrestrial vegetation. Such transitions are certainly part of the T-zone. However, the full suite of T-zone services indicates that the T-zone can be much broader in some settings. There is a relationship between topography, land use, runoff, T-zone services, and T-zone width that can be represented by a simple T-zone classification system. There is also a relationship between T-zone type and approaches to T-zone planning and management. These and other relationships are explained in sufficient detail to support the T-zone design and management recommendations provided later in this chapter.

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## DESCRIPTION OF TRANSITION ZONE

The T-zone provides the physical and ecological connection between the baylands and local watersheds. It connects the Bay to its developed as well as its undeveloped margins. It extends all along the bayshore and along the tidal reaches of rivers and streams. The T-zone includes the landward limit of the “backmarsh,” defined as the area between Mean Higher High Water and Extreme High Water or the Highest Observed

Tide (Ellis 1978, NOAA 2000)<sup>2</sup>. The T-zone extends uphill and upstream (i.e., landward) through the backmarsh to the limits of tidal effects on terrestrial and fluvial conditions. It extends downhill and downstream (i.e., bayward) through the backmarsh to the limits of the effects of terrestrial runoff and other freshwater discharges on conditions of the Baylands.<sup>3</sup>

The T-zone does not have a fixed width. It varies in width from place-to-place and over time. In the landward or upstream direction, the width of the T-zone is affected by several factors including the vertical range of the tide, the slope of the land, and the locations of built structures that control the upstream or landward movement of tidal water. In the bayward direction, the width of the T-zone mainly depends on the volume of terrestrial runoff entering the baylands. In general, for any given volume of runoff, the T-zone is wider where the tidal range is greater and where the land slopes gently to the bayshore. It is narrower where the tidal range is lesser and the land is steeper.

The width of the T-zone also varies at any given time and place based on the kinds and levels of ecosystem services it provides. For example, a broader T-zone is needed to provide marsh wildlife with refuge from high tides than if such refuge is not provided, and a broader T-zone is needed to accommodate sea level rise for the next century than for the next half-century. The relationship between T-zone width and the levels or kinds of its services is explained in more detail in the following section on ecosystem services of the T-zone.

### Landward Extent of the Transition Zone

The existing field studies of the T-zone (e.g., NOS 1975, Baye 2012, Thomson 2012, Beller et al. 2013, SFEI 2014) describe it somewhat differently, depending on the factors and processes being studied. When these studies are considered together, they suggest that the following field indicators can be used to estimate the maximum width of the T-zone at any location around the Bay. The indicators are presented in an order that reflects the distance landward of the backmarsh to which they pertain. The indicators most applicable to the landward margin of the backshore are presented first.

Some of these indicators must be applied in the field, and others can be applied using historical or modern maps and landscape imagery. However, all of these indicators are based on empirical observations. Each requires further development before it can be used in standardized ways by different practitioners to map or delineate the T-zone. Additional discussion of some of these indicators is included in the later section on T-zone mapping.

Not all the indicators are always applicable, due to both natural and unnatural factors. In urbanized settings, where the indicators might not be evident, best professional judgment can be used to apply these

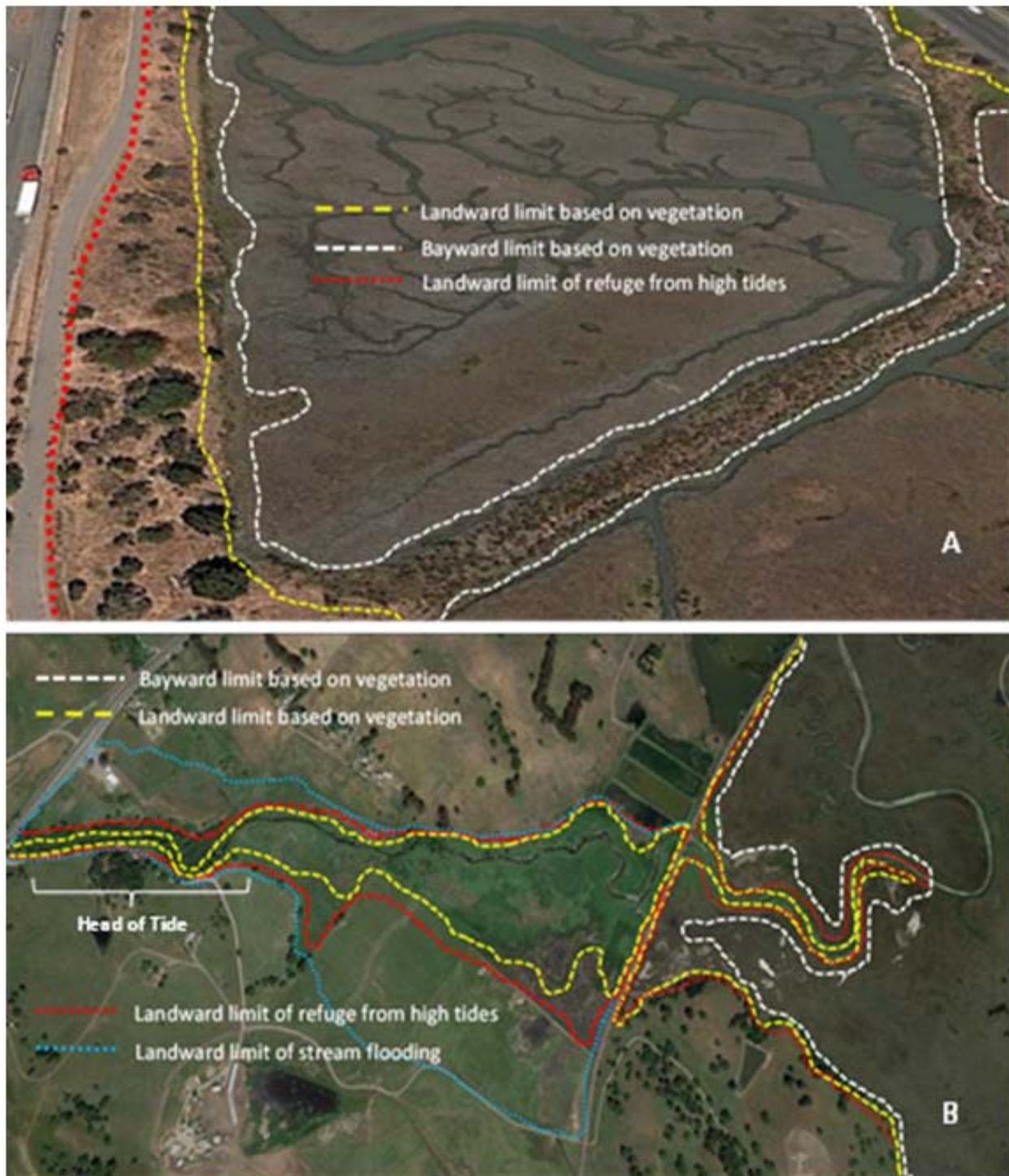
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<sup>2</sup> The landward boundary of the backmarsh can be difficult to delineate because it corresponds to the height of the uncommonly observed actual highest tide of the 19-yr metonic cycle (NOS 2000). The actual highest tide level is an “extratidal” water level because it is not included in tide height predictions (VIMS 2014).

<sup>3</sup> The baylands are the intertidal areas of the Bay (i.e., the areas between the elevations of the actual highest tide level of the 19-yr metonic cycle (NOS 2000) and the Mean Lower Low Water tidal datum), plus any adjoining areas that would be intertidal if not for the presence of levees, dikes, and other unnatural structures that artificially limit the landward excursion of Bay waters (Goals Project 1999).

indicators to historical maps and imagery to help determine the likely historical and potential future extent of the T-zone.

- *Tidal Marsh Vegetation.* One approach to determining the landward boundary of the T-zone is to identify the position of the backshore (Ellis 1978, NOAA 2000). This indicator is applicable to all the T-zone. The backshore is synonymous with the landward limit of the backmarsh as defined above. This approach is based on measuring where rooted tidal marsh vegetation becomes nonexistent in the landward direction along transects that are perpendicular to, and that cross over, the apparent Mean Higher High Tide contour (NOS 1975, Harvey et al. 1978). The T-zone that is delineated in this way is relatively narrow (see Figure 4.1A). It generally extends landward to an elevation less than 1m above the local Mean Higher High Water contour. However, there have been many observations of tidal marsh plant species occurring at higher elevations (e.g., Baye 2008), presumably due to occasional extreme high tides over many decades and perhaps also to the long-term aerial deposition of salts entrained in winds crossing Bay waters. Furthermore, the practice of depositing dredged saline sediments from the Bay onto levee tops and the historical side-casting of such sediments onto hillsides can result in the occurrence of salt-tolerant vegetation at unnaturally high elevations. These effects can broaden the T-zone when delineated using tidal marsh plants as indicators.
- *High Water Refuge.* The existence of dense plant cover above Mean Higher High Water that provides high tide refuge for tidal marsh wildlife can indicate where the T-zone exists landward of the backmarsh (see Figure 4.1A). This indicator can be used to determine if refuge is available. However, the preferred plant species and other structural characteristics of the refuge, including its width, will vary somewhat among the wildlife species to be served. Further development of this indicator is needed to specify these characteristics for key wildlife species. In most cases, vegetation does not have to be native or indicative of tidal influences. In other words, the wildlife refuge may exist landward of the tidal marsh vegetation.
- *Head-of-Tide.* The T-zone includes the head-of-tide (Figure 4.1B), which is defined through current studies (SFEI 2014) as the upstream limit of influences of tidal waters on channel geomorphology and hydraulics (Ensign et al. 2013, Florsheim et al. 2008, Bate 2002), aqueous salinity regimes and vegetation (Odum 1988, Brinson and Blum 1995), benthic sediment (e.g., van den Berg et al 2007, Ysebaert et al. 1993), benthic community composition (Strayer 2006, Kennish et al. 2004), and tide range (e.g., Gill and Schultz 2001). Owing to both daily and seasonal variability in tide heights (Marmer 1951), plus the variability in local river and stream flows, each HOT varies in width over time. The average width of each HOT depends on the slope of the channel bed, with steeper channels having narrower HOTs. Geomorphic, hydraulic and water quality conditions within the HOT are different than upstream and downstream conditions, such that the HOT supports unique plant and animal communities. A current study is concluding with a list of field indicators for identifying and delineating the HOT (SFEI 2014).
- *Habitat Mosaics.* The landward extent of the T-zone is influenced by complex interactions among tidal, fluvial, and terrestrial hydro-geomorphic processes that operate at the landscape scale. Collins and Grossinger (2004) and Beller et al. (2013) compiled written accounts of the landforms, habitat types, and vegetation patterns that were indicative of the landward extent of the historical T-zone of South Bay during the mid-nineteenth century (Figure 4.2). These historical data suggest that the salt marshlands were bordered in the landward direction by a band of salt-influenced vegetation, dominated by salt grass (*Distichlis spicata*), ranging in width from a few meters to more than a kilometer, depending on the steepness of the land. Near the backmarsh, the T-zone included freshwater springs and seeps, where near-surface fresh groundwater rose to the land surface over shallow intrusions of salt water (e.g., Harvey and Odum



**Figure 4.1.** Different T-zone boundaries corresponding to different ecosystem services, showing (A) the upper and lower boundaries based on plant species assemblages indicative of the backmarsh, plus the landward boundary of the high tide refuge service for the T-zone associated with a levee (Richmond, Contra Costa County); and (B) these same kinds of boundaries plus the range in Head of Tide (HOT) and the landward limit of the flood control service for the T-zone associated with a perennial stream (San Antonio Creek, Sonoma County). The area of stream flooding in (B) relates to a railroad grade that constricts the connection between the fluvial and intertidal portions of the floodplain.

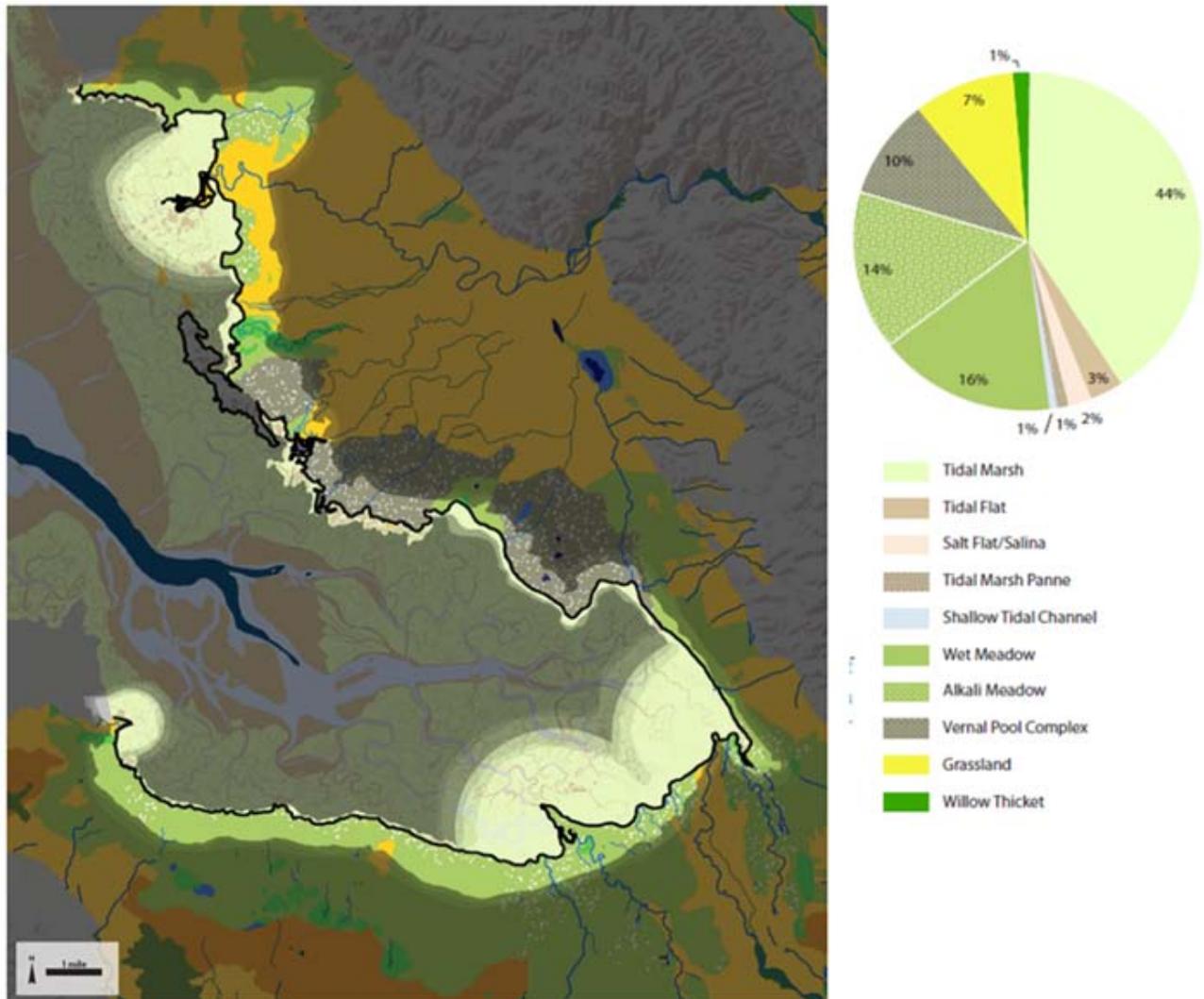
1990, Schultz and Ruppel 2002). Similar broad and complex habitat mosaics are also evident for brackish areas of the Estuary (Whipple et al, 2012). To better understand the nature of the T-zone and the full range of T-zone restoration potential, the studies recently completed for South Bay (Beller et al. 2013) and the Delta (Whipple et al. 2012) should be conducted throughout the rest of Estuary. Since most of the T-zone and adjoining uplands have been greatly modified by historical and modern land uses, there may be little existing evidence of the landward mosaics of T-zone habitats. As suggested above, these landscape-scale indicators can be applied to historical maps and imagery to estimate the potential for T-zone restoration.

- *Sea Level Rise Accommodation Space.* Planning the T-zone of the future must consider the likely landward migrations of the backshore and HOT due to sea level rise. If the T-zone becomes compressed between the rising Bay and steep natural lands or levees, its services will be diminished or lost completely. Therefore, broad areas for T-zone migration that can accommodate the full suite of local T-zone services in the future must be recognized as integral components of the existing T-zone. The width of the T-zone that is needed as accommodation space will depend on the rate and duration of sea level rise, the elevations and slopes of the lands and channels that are involved, and the presence of built structures that constrain the migration (Figure 4.3).
- *Ecological Connectivity.* In theory, the landward limit of the T-zone incorporates the movements of terrestrial wildlife to and from the baylands. The movements of small mammals and passerine birds suggest the T-zone extends 20-100m landward of the backmarsh (SFEI 2007). A wider T-zone has been noted for larger predators, such as coyotes and herons that have greater home ranges. Using wildlife movements to delineate the landward extent of the T-zone may not be practical at this time. However, local efforts to restore and protect the T-zone must consider how the particular needs of local wildlife that connect the T-zone to neighboring streams and terrestrial environments can be met through T-zone design and management. Once these needs are identified, the known natural history of the wildlife species of concern can be used to build ecological connectivity into T-Zone designs and management.

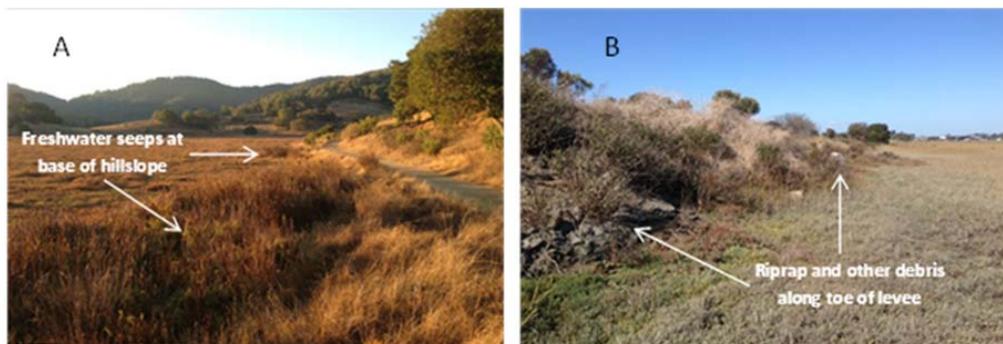
### Bayward Extent of the Transition Zone

An essential aspect of the T-zone is that it extends both bayward and landward from the backmarsh. It extends bayward through intertidal areas that are clearly distinguished by the effects of terrestrial runoff, intertidal emergence of fresh groundwater, and the effects of the deposition of terrestrial sediment and other materials on intertidal conditions. These effects increase the overall complexity of the T-zone and hence its biodiversity and resilience.

- **Bayward Extent of Backmarsh Processes.** Natural intertidal drainage processes and the activities of people within the backmarsh can affect the bayward extent of the T-zone. Along the bases of natural hillsides, the backmarsh commonly includes seasonal freshwater seeps. These can be prominent features in areas of abundant precipitation. They are indicated by patches of freshwater or brackish wetland vegetation in tidal marsh (Figure 4.3A). The backmarsh can also be subject to an accumulation of debris, sometimes referred to as the wrack line, deposited during high tides. Just landward of the wrack line (or contributing to it) can be eroded soils and fallen plant material originating from the adjacent areas above the backmarsh. Levees that provide public access to the backshore increase its visitation by people and pets that sometimes venture into the adjacent tidal marsh. The distance to which pets, especially feral or managed dogs and cats, can be expected to venture into tidal marsh is not well known. It is expected,



**Figure 4.2.** Map of the approximate historical extent of the transition zone (highlighted area) in South Bay (Beller et al. 2013). The transition zone is broadest bayward where intertidal vegetation is influenced by freshwater discharge from large streams. The transition zone is narrowest landward in the far southeast, where the adjoining land is steepest.



**Figure 4.3.** Examples of (A) freshwater and brackish vegetation indicating a freshwater seep along the backshore of saline marsh at China Camp, Marin County, and (B) riprap and other debris along the eroding bayward slope of an urban levee in Richmond, Contra Costa County.



**Figure 4.4.** Examples of pans that characterize the bayward habitat mosaic of the transition zone, showing (A) elongate pan at Whittell Marsh, Point Pinole, Contra Costa County; (B) diffuse pans in the area of tidal marsh landward migration into vernal pool east of Petaluma Marsh, Sonoma County; and (C) pans between hillsides and the landward limits of tidal marsh channels, China Camp, Marin County.

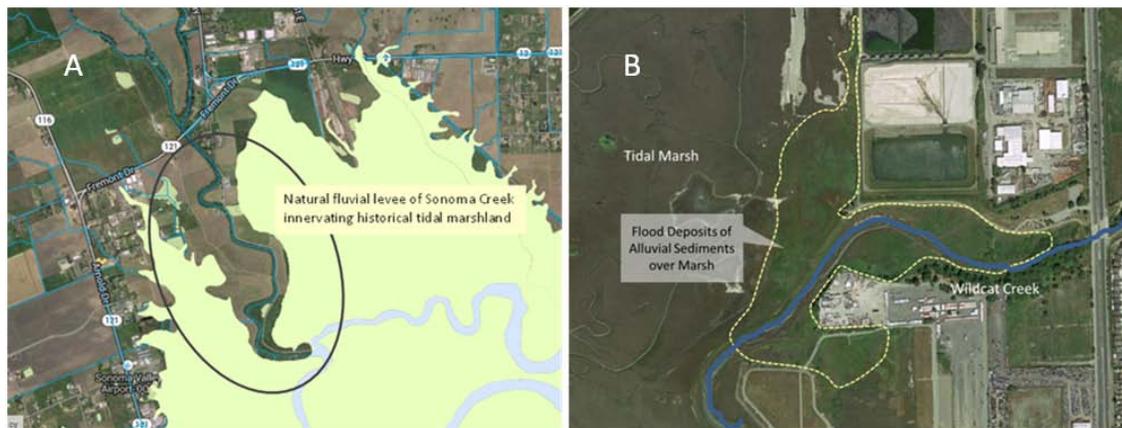
however, that pans can diminish the value of T-zone for wildlife (Simes 1999, Andrusiak 2003, Forrest and St. Claire 2006). Trash is a common component of the backmarsh wrack in urban settings (Figure 4.3B). In areas that are not affected by riverine discharges and that lack freshwater seeps, the backmarsh tends to develop distinctive habitat features and plant assemblages due to poor drainage and hypersaline soils. Elevated salinities are related to poor drainage. These flat intertidal areas are relatively high and landward of tidal marsh channels. The fine silts and clays deposited by high tides form dense, non-porous soils. Evaporation of the tidal waters trapped on the surface causes salts to accumulate. The combination of poor drainage and salt accumulation leads to the formation of shallow pans that lack vegetation (Figure 4.4). Once formed, these pans tend to persist as sites of salt concentration. The water in them can be brackish during the rainy season, when they trap precipitation, and hypersaline in the dry season.

- **Freshwater Discharge.** Freshwater discharge includes terrestrial runoff that reaches the T-zone through rivers, streams, canals and ditches, as well as non-saline effluent from water treatment facilities. The effects of freshwater discharge can be assessed as the bayward extent of tidal marsh plant species indicative of fresh or brackish water condition, and the bayward extent of fluvial bedload (i.e., the sediment that is transported by the discharge along the channel bed rather than in suspension). The extent of these effects is roughly proportional to the discharge volume, nutrient load, and total sediment bedload (see Figure 4.2 above). For any given volume of freshwater discharge, the extent of its effect on intertidal vegetation tends to be greater in brackish and saline areas of the Bay than in freshwater areas. That is, freshwater runoff has greater ecological effect in saline or brackish areas of the Bay than in freshwater areas. The yields of freshwater from larger watersheds and from sewage treatment facilities that discharge near the backmarsh can affect plant communities hundreds of meters bayward (H.T. Harvey and Associates. 2001 and 2002, Collins and Grossinger 2004, Grossinger et al. 2007, Hermstad et al. 2009, Grossinger 2009, Beller et al. 2010).

The bayward effects of terrestrial sediment loads can include the extension of fluvial levees into tidal marshes (Figure 4.5A), the deposition of sediments on marshes adjoining streams (Figure 4.5B), and the occurrence of brackish marsh vegetation in otherwise saline settings. The fluvial levees can serve as important avenues for the movements of terrestrial wildlife into and from intertidal areas. Trees on these levees are commonly used as roosts by birds of prey, including egrets, herons, hawks and owls that hunt in the baylands. Aggradation of channel beds (i.e., the build-up of the bed due to an accumulation of sediment

and other materials) in the T-zone can decrease the physical complexity of the bed and thereby also decrease its value as habitat, while increasing the risk of riverine flooding. Deposition of inorganic sediments on the tidal marsh plain can raise its tidal elevation, increase its bulk density, decrease its porosity, and thus cause shifts in the species composition of the marsh plant community (e.g., Byrd and Kelly 2006, Baye 2008, Palaima 2012).

Some studies of the effects of nutrient loads on the bayward reaches of the T-zone have focused on extreme nutrient enrichment of tidal marshes (e.g., Deegan et al. 2012). In these cases, nutrient enrichment may cause a lowering of the marsh plain by increasing the rate of microbial decomposition of organic soils. The effects of less extreme nutrient loads are likely to include increased vigor of marsh vegetation, as well as some shifts in plant community composition (Levine et al. 1998, Boyer and Zedler 1999, Hunter et al. 2008, Morris et al. 2013).



**Figure 4.5.** Examples of (A) a broad natural levee extending into former tidal marshland (Sonoma Creek, Sonoma County) and (B) a supratidal area caused by flood deposits of sediment on top of tidal marshland (Coyote Creek, Contra Costa County).

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## ECOSYSTEM SERVICES

Ecosystem Services are defined as the processes of ecosystems and their material and energy outputs that benefit people. Essential services provide such benefits as food and water, building materials and natural fuels, flood control and disease control, recreation and spiritual healing, pollution filtration, nutrient cycling, and biological diversity. In many cases, the values of the services are protected supported by laws and public policies.

The ecosystem services of the T-zone relate strongly to its role in connecting the baylands and their local watersheds (e.g., Ewel et al. 2001). This connection involves the conveyance of materials (e.g., the landward or bayward movement of water, sediment, and animals through the T-zone) and energy (e.g., the kinetic energy of moving water and the potential energy represented by the chemical bonds of the conveyed materials). The latter can be visualized as the energy released through the decomposition of trash and plant detritus that enters the T-zone from the Bay or from the local watershed. Much of the food web of the intertidal portion of the T-zone is probably based on detritus (Mitsch and Gosselink 2007).

The importance of the functional relationships between the T-zone and local watersheds should be emphasized. Most of the inorganic sediment that accounts for the formation and persistence of the tidal

marshes is derived from local watersheds (McKee et al. 2013). As mentioned above, the freshwater runoff from local watersheds creates salinity gradients through the baylands and into the Bay that greatly increase the overall biodiversity of the region. Many species of terrestrial and aquatic wildlife, including birds of prey and salmon between the Bay and its local watersheds move through the baylands. In many regards, the Bay and its local watersheds are linked together by the baylands, and the mechanisms of this linkage are the workings of the T-zone.

The T-zone has always been the connection between the Bay and its watersheds. It has always supported the same basic kinds of natural processes and played the same natural roles. How people have valued them as services has changed, however. They are more valued today, due in part to better scientific understanding of their benefits to society. A general description of the most recognized T-zone services is presented here.

### Ecological Resilience

For the purposes of this chapter, ecological resilience is defined as the amount of disturbance that an ecosystem can withstand without undergoing major changes in its stable state, as measured by its form and structure, or the time an ecosystem takes to return to a stable state following perturbation (Holling 1973, Gunderson 2000, Walker et al. 2004). Resilience therefore varies from ecosystem to ecosystem and over time. One challenge in restoration ecology is to understand the natural self-organizing processes that lead to resilience. A major goal of BEHGU is to increase the ecological resiliency of the baylands as a whole, and it is assumed that this depends on restoring and sustaining a resilient T-zone. Providing upland buffers and accommodation space for estuarine migration, as well as assuring that Baylands designs are consistent with natural processes of Baylands formation and maintenance can help achieve resilient Baylands ecosystems.

There is a theoretical relationship between the ecological complexity of ecosystems, their overall resilience, and the success of their management (Folke et al. 2004, Holling 2004, and Campbell et al. 2009). In general, it is expected that management success increases with resiliency, which increases with complexity. Therefore, increasing the ecological complexity of the T-zone is assumed to be one strategy for increasing its resilience, as well as that of the baylands ecosystem. High levels of all of the following services are assumed to be consequences of a resilient T-zone.

Providing ecological resilience for the baylands ecosystem is regarded as an over-arching service of the T-zone. The levels of all its other services can be used to assess its own resilience and its contribution to the resilience of the Baylands.

### Buffering

The T-zone serves as a buffer in two basic regards. It moderates the landward effects of tidal processes and it moderates the bayward effects of fluvial and terrestrial processes. Most of the buffering relates to controlling pollution, erosion, and flooding.

- **Pollution Control.** The vegetated portions of the T-zone can help improve the quality of tidal waters by filtering organic and inorganic pollutants, including fine sediment, from terrestrial runoff and tidal flood waters. The trapped sediment can serve to bind and sequester heavy metals and nutrients (Odum 1990).

The T-zone can include the portions of tidal marshes that are intentionally used to help treat urban runoff and effluent from sewage treatment facilities.

- **Biological Invasion Control.** If the T-zone is densely vegetated with native plant species, it can help protect adjoining intertidal habitats from invasion by non-native terrestrial vegetation by creating a barrier to their bayward dispersal (e.g., Woolfolk 1999; Baye 2008; Fetscher et al. 2010; Wasson and Woolfolk 2011).
- **Erosion Control.** Vegetated intertidal habitats help protect adjoining terrestrial habitats by reducing their risk of erosion (Seliskar and Gallagher 1983; Garbisch and Garbisch 1994; BCDC 2011). This service of the T-zone mostly occurs during very high tides that inundate the marsh plain so deeply that the marsh does not effectively the erosive energy of ship wakes and wind-generated waves.

### Flood Control

The T-zone can help reduce the hazards of riverine flooding by if it includes the following landscape features or elements.

- **Channels.** The tidal reaches of channels that convey riverine floodwaters or effluent to the Bay are part of the T-zone, to the extent that the waters measurable affect marsh plant community structure.
- **Floodplains.** The landward portions of intertidal floodplains that disperse floodwaters and the riverine floodplains bordering the Head-of-Tide (HOT) are part of the T-zone. The high marsh plain can also serve to attenuate wakes and wind-generated waves that otherwise are more likely to erode levees that prevent tidal flooding.
- **Floodwater Storage Areas.** Diked baylands and natural or artificial depressions on riverine floodplains bordering HOT that store floodwaters are part of the T-zone.

### Sea Level Rise Accommodation

This is a complex service relating to sustaining intertidal habitats, especially tidal marsh and the tidal reaches of rivers and streams, as well as appropriate bayshore land uses, such as flood water dispersal and storage that are threatened by accelerated sea level rise (see Science Foundation Chapter 3).

Accelerated sea level rise threatens critically important infrastructure and many land uses around the Bay (BCDC 2011). The need to plan for the future T-zones provides opportunities to examine alternative management responses to this increasingly serious threat. The undeveloped areas that are landward of the existing T-zone can provide ways for the T-zone to migrate landward and upstream with minimum social impacts.

Sea level rise causes the intertidal zone and the T-zone to migrate landward and upward into local watersheds (e.g., Wasson et al. 2013). The leading edge of this migration is usually marked by the landward margin of new backmarsh. Four factors mainly influence the width of resulting marshland: the rate of sea level rise, the steepness of the land surface across which the tidal marsh can migrate, the amount of fine inorganic sediment (i.e., silts and clays) deposited on the marsh plain during tidal and riverine flooding, and the amount of organic materials (i.e., roots and plant litter) that are produced by the marsh vegetation and

contribute to the height of the marsh plain. Increases in the supply of organic and inorganic sediment can offset increases in the rate of sea level rise (e.g., Mudd et al. 2009). To some degree, sea level rise can be regulated with water control structures that are built into levees and across creeks. However, these structures can be expensive to build and maintain, and they can have unintended biological and water quality impacts. For any set of these factors, the potential width of the new marshland is mainly controlled by the topographic slope immediately landward. By providing gently sloping lands across which the T-zone can migrate, the resilience of the marsh ecosystem to sea level rise is greatly enhanced.

Sea level rise and changing precipitation patterns will drive geomorphic changes at the heads-of-tide (HOT - see description of HOT and references therein in the previous section about the bayward extent of the T-zone). By providing naturalistic stream gradients and ample floodplains upstream and adjacent to existing HOTs, their services can be conserved and the hazards of flooding can be reduced.

### Nutrient Processing

Much of the T-zone consists of wetlands (see profiles of T-zone types in Appendix 4.2). All wetlands tend to have high rates of nutrient assimilation. They also tend to have very high rates of primary productivity, and they tend to retain much of the organic material they produce. The terrestrial slope wetlands that exist along the backmarsh undoubtedly provide nutrients downslope to the adjoining intertidal habitats. This is commonly indicated in the field by greater vigor or stature of tidal marsh vegetation along the immediate landward boundary of the backmarsh. It has also been hypothesized that tidal marshes export nutrients to coastal waters, where they subsidize estuarine food webs (*Valiela and Teal 1979*, Odum 1980). There have been many studies to understand the direction and magnitude of nutrient fluxes to and from tidal marshes, as well as the underlying mechanisms (Childers et al. 2002; Nixon 1980; Odum 2002; Stevenson et al. 1988). The results of these studies are equivocal. While it is certain that tidal marshes tend to be net nutrient producers (Hammer 1989, Tiner 2013), whether or not the net direct of nutrient flux is bayward may vary from one marsh to another, and over time.

### Groundwater Recharge

The areas of the T-zone that include riverine floodplains or stormwater retention basins can help maintain near-surface groundwater levels through recharge during floods. Such recharge constrains near-surface saltwater intrusion, while maintaining freshwater slope wetlands, springs, and seeps in or near the backmarsh. If the groundwater is high enough it can discharge into tidal reaches of streams within the T-zone during low tide, thus maintaining downstream salinity gradients that increase biological diversity.

### Biological Diversity Support

Broad ecological transitional zones tend to be biologically diverse (e.g., Naiman and Dé camps 1990, *Karika and van Rensburg 2006*). They tend to be areas where biological communities or assemblages overlap, and they can have their own endemic flora and fauna, (Odum 1953, Holland 1988, Holland et al. 1991, Harding 2000, West and Zedler 2000, Leppig and While 2006, Senft 2009). The Bay Area T-zone supports a variety of plants and animals of special management concern (Table 4.1).

The biological diversity of the T-zone is the consequence of many ecological processes. The key processes are discussed below. Additional information about the value of the T-zone to wildlife is presented in Science Foundation Chapter 5.

- Wildlife Refuge and Predation.** The T-zone provides refuge from both physical and biological sources of stress and mortality for wildlife species. For example, it serves as a refuge from tidal and riverine flooding for both intertidal and upland biota, including numerous rare and endangered species (Chapman et al. 1996, Sedell et al. 1990, Semlitsch and Bodie 2003, BCDC 2011, Josselyn 1983, Goals Project 1999, 2000). During major flood events, tidal marsh wildlife tends to be concentrated in the T-zone, which therefore can serve as an important foraging area for many species of predators. The degree to which the T-zone serves as refuge is likely to be proportional to the width of the T-zone and its structural complexity.
- Wildlife and Plant Movement.** The T-zone supports the migration and dispersal of plant and animal species. It enables them to move along the bayshore between patches of preferred baylands habitat. For example, the endangered California clapper rail and salt marsh harvest mouse use the T-zone to move between patches of tidal marsh that are otherwise discontinuous (Fisler 1965, Botti et al. 1986, Shellhammer 1989, Overton 2014). The T-zone will also enable baylands wildlife to track or avoid changes in salinity due to future sea level rise. There are many species of wildlife that regularly travel into the T-zone from adjacent terrestrial areas to forage (SFEI 2007). Where the T-zone involves a river or stream, it can support the seasonal movements of anadromous fishes and other wildlife between the Bay and local watersheds.
- Evolutionary Adaptation.** The survival of local populations of plants and animals depends on their adaptation to changes in habitat conditions. Such adaptation is known to occur at the margins of habitats, including in ecotones where individuals encounter the limits of their physiological tolerance to environmental factors (Mayr 1970, Lesica and Allendorf 1995, Schilthuizen 2000, Gaston 2003, Karka and van Rensburg 2006). For some species, the T-zone may be critically important as a place for adaptations to changes in habitat conditions caused by sea level rise. For example, it can be hypothesized that the ability of larvae of coastal populations of the red-legged frog (*Rana draytonii*) to tolerate brackish water salinities reflects adaptation to increasing salinities at breeding sites within the T-zone (Collins and Collins 2007), given that the general intolerance of frogs to salinity is well known (Ruibal 1959), and other coastal species of frogs have shown adaptation to fluctuating intertidal salinity (Gomez-Mestre and Tejedo 2003, Rios-López 2008). Similarly, the adaptation of brackish-water plants to increased salinity could be occurring in the T-zone of brackish marshes, where soil salinities tend to be locally elevated.
- Landscape Complexity.** The T-zone contributes to a complex mosaic of bayland habitat types (see discussion of habitat mosaics in the above section on factors affecting the landward extent of the T-zone) that increase the local diversity and abundance of plant and animal species across landscapes at a regional scale (Poiani et al. 2000, Moritz 20002, Huber et al 2010; also see Chapter 6).

**Table 4.1.** Native wildlife and plant species of concern for management of terrestrial-tidal marsh ecotones (from Baye 2008).

Species	Regional Habitat and Range
<b>Ridgway's rail</b> ( <i>Rallus longirostris obsoletus</i> )	Tidal salt marsh, tidal brackish marsh: SF Bay, San Pablo Bay, western Suisun Marsh and Martinez marshes.
<b>California black rail</b> ( <i>Laterallus jamaicensis coturniculus</i> )	Tidal brackish marsh (occasionally salt marsh): Suisun Marsh, Martinez Marshes, San Pablo Bay; local in SF Bay.

<b>Virginia rail</b> ( <i>Rallus limicola</i> )	Tidal brackish marsh, non-tidal brackish or freshwater marsh; throughout Estuary.
<b>Salt marsh harvest mouse</b> ( <i>Reithrodontomys raviventris</i> )	Tidal or non-tidal salt or brackish marsh, middle and high marsh zone; abundant pickleweed; throughout Estuary.
<b>Salt marsh wandering shrew</b> ( <i>Sorex vagrans halicoetes</i> )	Tidal salt marsh, middle marsh zone, abundant invertebrate prey, driftwood; San Francisco Bay south of Golden Gate
<b>San Francisco Estuary song sparrow subspecies</b> ( <i>Melospiza melodia</i> spp)	Tall high tidal marsh vegetation near tidal creeks and adjacent terrestrial scrub.
<b>San Pablo vole</b> ( <i>Microtus californicus sanpabloensis</i> )	Tidal marshes around the mouth of San Pablo Creek (Contra Costa County)
<b>Suisun shrew</b> ( <i>Sorex ornatus sinuosus</i> )	Tidal salt or brackish marsh, northern San Pablo Bay, Suisun Marsh, dense low vegetation with woody debris
<b>Red-legged frog</b> ( <i>Rana draytonii</i> )	Occasionally breeds in brackish marsh backshore pools
<b>Soft bird's beak</b> ( <i>Cordylanthus mollis</i> ssp. <i>mollis</i> )	Tidal brackish or salt marsh, high marsh zone with sparse, low cover; northern San Pablo Bay to Suisun Marsh, Martinez Marshes
<b>Northern salt marsh bird's-beak</b> ( <i>Cordylanthus maritimus</i> ssp. <i>palustris</i> )	Tidal salt marsh, high marsh zone with sparse, low cover; Sausalito to Petaluma River (Marin County)
<b>Suisun thistle</b> ( <i>Cirsium hydrophilum</i> var. <i>hydrophilum</i> )	Tidal brackish marsh, middle to high marsh zone, Suisun Marsh
<b>Salt marsh owl's-clover</b> ( <i>Castilleja ambigua</i> ssp. <i>ambigua</i> )	Tidal salt or brackish marsh, high marsh zone, San Pablo Bay and Suisun Marsh (historic range also in SF Bay)
<b>California sea-blite</b> ( <i>Suaeda californica</i> )	Tidal salt marsh bordering sandy beaches, Central and South San Francisco Bay (historic range)

### Cultural Support

Cultural support refers to the importance of the T-zone to regional and local human history, and to its value for environmental science, education, recreation, and spiritual healing.

The transition zone provides most of the public access to the Bay. For example, the popular Bay Trail traverses much of the T-zone. Most recreational fishing in the Bay occurs at piers and along the backmarsh of the T-zone. Environmental science, education, and recreation are major aspects of the region's culture, and much of this involves the T-zone.

Ongoing research in the T-zone is focused on such fundamentally important topics as the effects of sea level rise on nearshore habitats, the effects of human visitation on wildlife, and how coastal engineering might mitigate the hazards of sea level rise. Numerous environmental education programs use the T-zone as an outdoor laboratory. Many non-governmental environmental organizations (e.g., Students and

Teachers Restoring A Watershed (STRAW), Save The Bay, Marin Audubon Society, Acterra) work to clean the T-zone of trash, and to restore the T-zone's native plant communities.

This intensive public interest in the T-zone is not new. There are many cultural resources representing prehistoric to historical communities of peoples that are associated with the T-zone. These include ancient shellmounds and other evidence of Indian occupation and use, the embarcaderos of the mission period and gold rush, the remnants of railroads and Chinese fishing villages of the early industrial period, and historical ranching, dairying, and military buildings. The T-zone retains important evidence of many distinctive cultures in the region.

### Carbon Sequestration

Wetlands are important in the global carbon balance (Mitra et al. 2005, Bridgham et al. 2006, Crooks et al. 2014). They serve as important carbon sinks, due to their fast rates of primary productivity, large standing biomass, and tendency to retain much of this material in the form of highly organic sediments (Zedler and Kercher 2005). The amount of carbon sequestered in the T-zone is probably proportional to the amount of the T-zone that is comprised of wetlands. Carbon sequestration in the baylands is not limited to the T-zone, however, and it is covered in more detail in Science Foundation Chapter 6.

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## TRANSITION ZONE TYPOLOGY

Based on the existing studies of the historical T-zone (Collins and Grossinger 2004, Beller et al. 2013), field surveys that characterize its present-day condition (NOS 1975, Baye 2012, Thomson 2012, Beller et al. 2013, SFEI 2014), and the current scientific understanding about how the T-zone is formed and naturally sustained, a T-zone typology in two parts has been developed. One part organizes the T-zone into types based on formative processes and physical structure. The second part organizes the T-zone into Sub-zones based on the different spatial limits of its ecosystem services.

### Types of Transition Zones

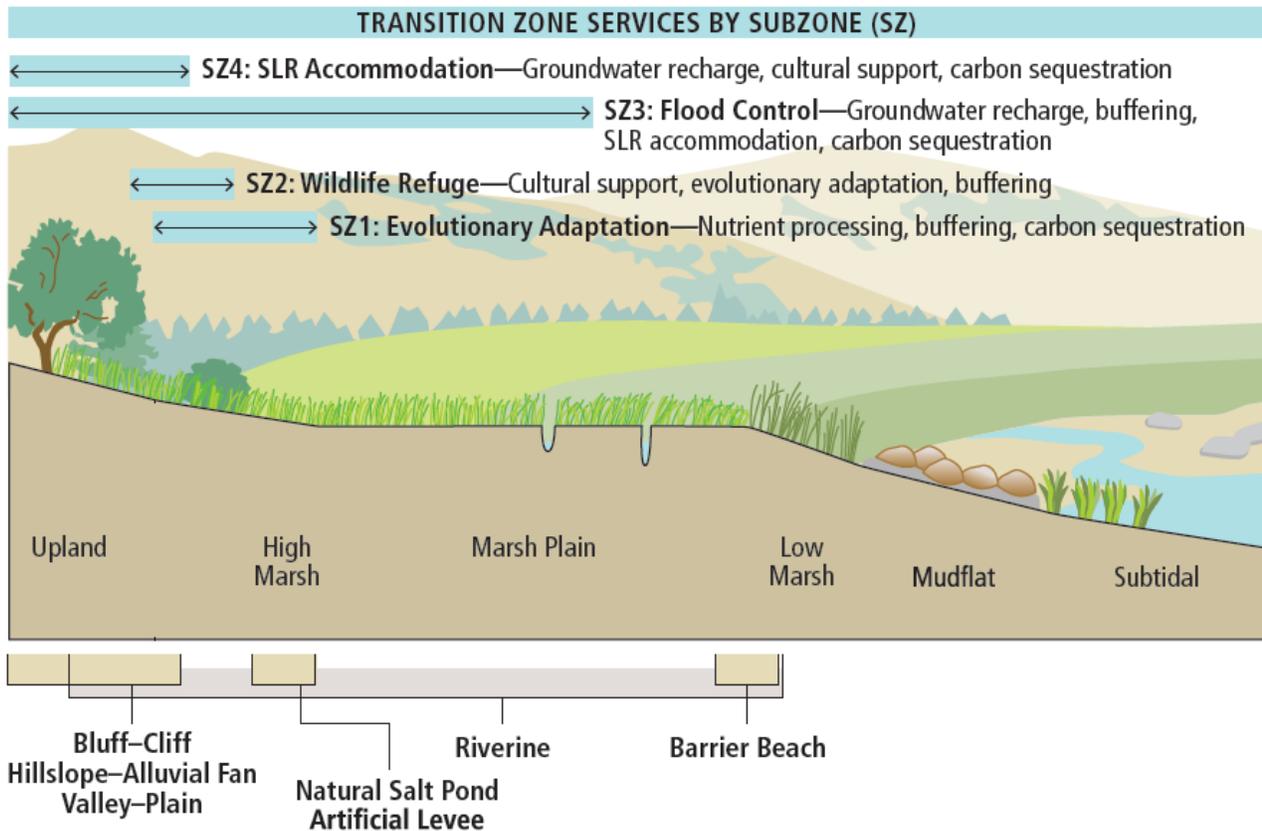
Seven T-zone types have been identified based on differences in the environmental factors and processes that govern T-zone formation (Table 4.2). Each type is profiled in detail in Appendix 4.2. In aggregate, these seven types of T-zone represent the full range of existing T-zone conditions for the Estuary downstream of the Delta.

The typology can serve to guide T-zone restoration and management. For example, successful restoration will require knowing what type of T-zone is best suited for a given restoration site, based on the local controlling factors and processes. Mismatches between T-zone types and settings will cause restoration efforts to fail expensively.

As explained in Appendix 4.2, one type of T-zone, the Barrier Beach type, is usually dissociated from the backmarsh and HOT. It often occurs at the bayward margin of tidal marsh. It is identified as a type of T-zone because it provides many of the same ecosystem services as the other T-zone types. For example, Barrier Beaches can serve as high tide refuge and they support evolutionary adaptation and movement of intertidal plants and animals.

### Sub-zones of the Transition Zone

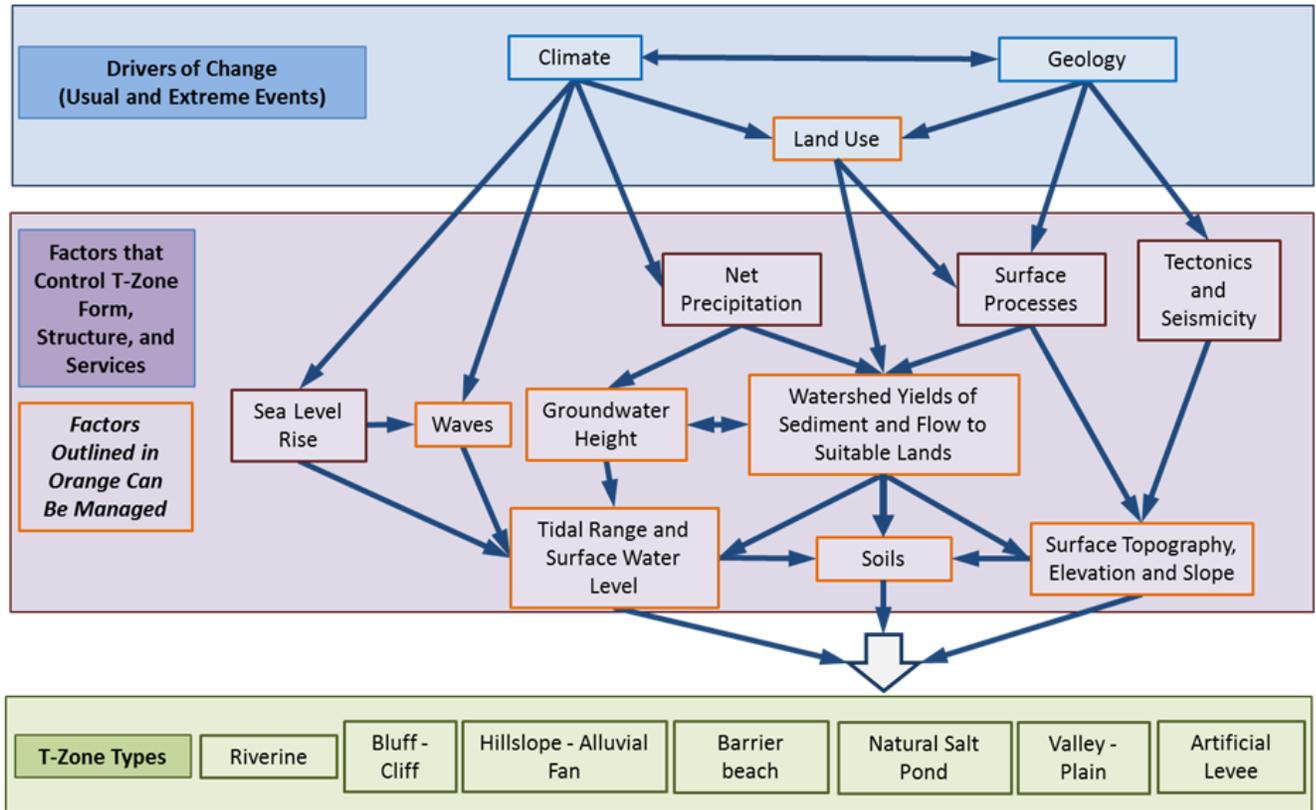
Each type of T-zone can be sub-divided into 2-4 Sub-zones (SZ) based on the different extents (i.e., “footprints”) of key ecosystem services (Figure 4.6). The levels of service provided by each Sub-zone vary among the T-zone types. There are many factors that influence the levels of services provided by any particular Sub-zone (see Figure 4.7). But, in general, the wider and less disturbed sub-zones provide higher levels of their characteristic services (Table 4.2).



**Figure 4.6.** Diagram of spatial relationships among T-zone types, Sub-zones, and tidal datums of the baylands ecosystem. Sub-zones SZ# and SZ4 extend landward of the upland shown in this figure. The Riverine Type extends bayward to the limits of the effects of freshwater discharge on the intertidal vegetation. The primary services of each Sub-zone are shown in bold. Ubiquitous services (e.g., ecological resilience, animal movement, landscape complexity) are not shown because they do not help differentiate the Sub-zones.

The differentiation of the T-zone into Sub-zones helps to organize efforts to achieve specific kinds or levels of services through T-zone design and management. In other words, the levels of different services can be controlled to some degree by the design and management of the Sub-zones. One step in T-zone planning or restoration is to decide what type of T-zone is best suited to the restoration site, and another step is to decide how wide each Sub-zone should be to provide needed kinds and levels of service (see section below on T-zone planning and management).

For convenience, the Sub-zones are numbered in the order they would be encountered by a person walking landward from the backmarsh. For example, anyone walking uphill from the landward edge of a tidal marsh would first encounter Sub-zone 1 (SZ1). Assuming that the T-zone has an area of dense vegetation



**Figure 4.7.** Conceptual model of the effective relationships among drivers and factors affecting type, condition, and ecosystem services of the transition zone. Factors that can be managed to achieve desired transition zone conditions are outline in orange.

above the backmarsh that serves as wildlife refuge, the person would encounter Sub-zone 2 (SZ2). Walking further uphill or landward from the backmarsh the person might encounter an area of diked marsh used to store riverine floodwaters or to buffer the refuge and tidal marsh. This would be Sub-zone 3 (SZ3). Walking further landward the person would be entering Sub-zone 4 (SZ4), which is the area dedicated to sea level rise accommodation.

People who rarely visit any part of the bayshore except via a trail atop a bayshore levee might understandably regard the levee as the entire T-zone. Assuming that the levee supports the kind of vegetation needed to provide some amount of wildlife refuge, and that no floodwaters are stored behind the levee, and that there is no possible accommodation space further landward, the T-zone is effectively

**Table 4.2.** Expected kinds and levels of ecosystem service for T-zone types and their Sub-zones (dark green = high level of service; light green = medium level; white = low level). Indicated levels of service are generalities and can be adjusted through T-zone design and management (e.g., see table footnotes pertaining to SZ3 of Riverine Type and SZ 4 of Levee Type).

Ecosystem Service		T-zone Sub-zones and T-zone Types			
		Sub-zone 1 (SZ1) Endemism	Sub-zone 2 (SZ2) Refuge	Sub-zone 3 (SZ3) Flood Control	Sub-zone 4 (SZ4) Sea Level Rise Accommodation
		All Types	All Types	Riverine <sup>A</sup>	Hillslope-Fan Valley-Plain Riverine Levee <sup>B</sup>
Buffering	Pollution Control	H	M	M	L
	Non-native Invasion Control	M	M	L	M
	Erosion Control	H	H	L	L
Flood Control		L	L	H	L
Sea Level Rise Accommodation		L	L	H	H
Nutrient Processing		H	L	M	L
Groundwater Recharge		L	L	H	H
Biological Diversity Support	Wildlife Refuge and Predation	H	H	M	M
	Wildlife and Plant Movement	M	H	M	M
	Evolutionary Adaptation	H	M	L	L
	Landscape Complexity	M	M	M	H
Cultural Support		L	H	L	H
Carbon Sequestration		H	L	L	M
<p><sup>A</sup> Includes diked baylands used to store floodwaters.</p> <p><sup>B</sup> Sea Level Rise Accommodation can be enhanced for the Levee Type of T-zone by greatly reducing the slope of the side of the levee facing the Bay to create uplands for the Bay to migrate onto. This concept has been termed the “horizontal levee” (The Bay Institute 2013).</p>					

restricted to SZ1 (i.e., a narrow area along the bayward levee face), SZ2 (a narrow area of wildlife refuge on the levee face and perhaps atop the levee), and a bayward component of SZ3 (the tidal marsh plain that attenuates wave energy). The T-zone in this case would probably support evolutionary adaption along the backmarsh at the base of the levee, provide some refuge from high tides, attenuate waves to some degree, and provide some amount of cultural support in the form of passive recreation (e.g., hiking, jogging, and bird watching along the levee). The levels of these services could be significantly reduced by the narrowness of the sub-zones, their lack of physical complexity, and prevalent visitation by people and terrestrial predators (Foerster and Takekawa 1991, Baye 2008). These considerations notwithstanding, the

services provided by the Levee Type of T-zone can be locally very important to the health and safety of people and wildlife.

SZ2 deserves special consideration because of its value as a high tide refuge for residential birds and small mammals, some of which are rare or endangered. Furthermore, it supports a unique plant community adapted to very infrequent tidal flooding that includes several rare, threatened, and endangered plant species. It generally extends from just above the local Mean Higher High Water (MHHW) contour to the ecologically effective limit of tidal influence, assumed to be the locally Highest Observed Water Level (HOWL) (Harvey et al 1978, NOS 1978, Thomson 2013, Fulfrost and Thomson 2014). Using these tidal datums to bound SZ2 can facilitate its mapping. SZ2 is especially sensitive to invasion by non-native plants and excessive visitation by people. In the current landscape, SZ2 is one of the most threatened features of Pacific coast tidal salt marshes (Traut 2005). Bay Area losses of natural SZ2 have been estimated to be 90% (Shellhammer 1982), and the ecological services of the remaining areas of SZ2 have been severely degraded (Baye 2004). More information on SZ2 particular to transitions between tidal marsh and upland is in Appendix 4.3.

The stratification of the T-zone into a number of contiguous sub-zones based on the spatial distributions or “footprints” of its services has precedent in the design and management of riparian buffers. Many public agencies responsible for riparian buffers subdivide them into three or more component zones that correspond to different kinds or levels of buffering (e.g., Wenger 1999, Collins et al. 2006, Johnson and Buffler 2008). The riparian definition recommended by the National Research Council recognizes that the ecological functions or services indicative of riparian areas tend to extend different distances from the adjoining bodies of water (NRC 2002). The T-zone as defined here, absent sub-zone SZ4, is generally consistent with the NRC riparian definition. From the perspective of riparian science, the T-zone as defined here is essentially the riparian zone of the Bay.

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## **TRANSITION ZONE MAPPING**

None of the efforts to map the tidal wetlands of the Bay Area (Jones and Stokes Associates et al. 1979, Dedrick 1989, Dedrick and Chu 1993, SFEI 2011) have explicitly identified any part of the T-zone. However there has been some effort dedicated to mapping the Bay Area T-zone alone. Two such projects have explicitly identified parts of the historic (Beller et al. 2013), current and future (Fulfrost and Thomson 2014) T-zone. Important regional efforts to map non-aquatic terrestrial habitats and land use, such as the Conservation Lands Network (<http://www.bayarealands.org/about/>), the head-of-tide (HOT) mapping project (SFEI 2014), the California Vegetation Classification and Mapping Program (VegCAMP) (<http://www.dfg.ca.gov/biogeodata/vegcamp/system>), and the National Land Cover database (Collin et al. 2012) have also not explicitly identified the T-zone. Efforts to map floodplains (FEMA 2003), flood infrastructure (SFEI 2013), the riparian zone of tidal marshlands (SFEI 2010), and to predict the effects of climate change on future Bay water levels (Veloz et al. 2012, NOAA 2012), will provide essential data for mapping the T-zone, but will not by themselves or in combination provide a comprehensive regional T-zone map, based on the definition and classification of the T-zone recommended here.

Ecological transitions are challenging to map because of the dynamic nature of their formative processes that result in abundant variability from place to place and over time. The challenge is made more difficult by the need to resolve the complexity of the T-zone into a practical map that planners and managers can use to track local and regional changes in T-zone extent and condition, and to prioritize T-zone restoration and protection needs and opportunities (Fulfrost and Thomson in preparation, SFEI 2014).

The optimal approach to mapping the T-zone will probably involve estimating the extent of each type of T-zone and the width of their Sub-zones. This will facilitate linking the map to expected ecosystem services and management actions. Furthermore, there should be guidelines for mapping the T-zone at two spatial scales: local (project-specific), and regional (i.e., across multiple projects within a subregion or for the Bay as a whole). The two spatial scales of mapping are needed to assess the local performance of projects as well as their cumulative impacts on T-zone extent and condition. A regional map is especially important for prioritizing restoration and protection efforts, assessing the relative effects of projects and ambient climate change, and evaluating the efficacy of state and federal policies used to govern land use affecting the T-zone.

### Regional Scale

Mapping efforts at the regional scale can be completed with GIS-based techniques using, as much as is feasible, existing datasets. It is also possible to manually map from aerial imagery or even conduct surveys. However, the time and expense of these approaches can limit the ability of practitioners to employ them across broad geographic areas. Efforts to comprehensively map the T-zone across the region need to be developed for a variety of reasons, including but not limited to the following: the T-zone cannot be protected if its existing and likely future extents are unknown; a map of the T-zone is needed to guide assessments of its condition, understand baseline conditions for assessing future change, and to prioritize conservation actions.

Much work is needed to identify the best indicators and supporting data for mapping the T-zone types and their service-based Sub-zones. The previous section describing the landward and bayward extents of the T-zone present some of the empirical field indicators that could be further developed to support accurate T-zone mapping. Any regional efforts to map the T-zone should be coordinated with efforts to map other landscape elements, such as wetlands, streams, flood-prone areas, and flood control infrastructure. A plan for such coordination is needed. In support of such a plan, the alignment between existing regional mapping efforts and the need for a regional T-zone map has been explored (Table 4.3). This is only an initial look - it should only serve as a starting place for a more thorough assessment of ways to coordinate and advance a regional T-zone mapping effort.

Of the various mapping efforts reviewed, several stand out as essential components of a regional T-zone map. These three efforts together will greatly increase the capacity of environmental planners and managers to protect the ecosystem services of the T-zone. It must be emphasized, however, that there is, at this time, no concerted effort to integrate these efforts or others into a comprehensive methodology for mapping the T-zone.

Landward Extent of the effects of tidal processes on vegetation (Fulfrust and Thomson, in preparation). This is a regional effort to map the likely current landward extent of tidal marsh vegetation, as affected by extreme high tides and wave run-up. When combined with existing maps of tidal marshes, flood infrastructure, and HOT (i.e., SFEI 2011 and 2013, SFEI 2014), this map will depict the habitat composition and topography of the landward portion of SZ1 (i.e., the portion of SZ1 that is landward of local Mean Higher High Water), and the bayward portion of SZ3 (i.e., the tidal marsh plain for distributing riverine flood waters at low tide and for attenuating waves).

- **Landward Extent of Tidal Flood Risk (NOAA 2012).** The purpose of this model and its online visualization tool is to provide coastal managers and scientists with a preliminary look at potential sea level rise and coastal flooding hazards. The viewer is a screening-level tool that uses nationally consistent data sets and analyses. The data and maps can be used at several scales to help gauge trends and prioritize actions for different sea level rise scenarios. This model and its output maps could be augmented with other existing and forthcoming regional maps that more accurately depict topography (e.g., OPC 2012), hydrography (SFEI 2011), flood infrastructure (SFEI 2013), and land use (e.g., Collin et al. 2012, Conservation Lands Network <http://www.bayarealands.org/about/>) to better estimate the landward extent of SZ4.
- **Head-of-Tide (SFEI 2014).** An existing study will establish a protocol for mapping the HOTs around the Bay, starting with a pilot project on six local streams (SFEI 2014). The resulting protocol will allow multiple practitioners to estimate the location of HOT for any natural or artificial drainage channel connected to the intertidal area of the Bay. When combined with existing maps of tidal marshes, levees, and flood-prone areas (FEMA 2003), this map will depict the extent of the landward component of SZ3 (i.e., terrestrial areas and diked baylands used to disperse or store riverine floodwaters). This composite map could be combined with the map of tidal marsh (SFEI 2013) to visualize the approximate entire extent of SZ3.

### Local Scale

Local mapping refers to the development of maps to inform local planning and management of T-zone restoration projects. Project-specific maps can also be used to analyze levels of ecosystem services and their controlling processes, to assess the regional diversity and extent of the T-zone, and to track the effects of sea level rise that are not detectable at coarser scales. High-resolution local re-mapping of the T-zone can be used for early detection of sea level rise impacts on T-zone conditions. Greater map resolution and accuracy are needed to support these local purposes of T-zone mapping.

There is currently no single accepted method for high-resolution local mapping of the T-zone, as defined here. It is likely that the optimal methodology will involve validation through field delineation. Trial use of remotely sensed vegetation indicators to estimate the landward and/or bayward limits of SZ1, SZ2, and SZ3 (See Table 4.3) suggest that vegetation might also be used to delineate these Sub-zones on the ground. As noted above, vegetation indicators have been used to delineate both the landward and bayward limits of SZ1 and SZ2 (e.g., NOS 1975, Harvey et al 1978, H.T. Harvey and Associates 2002, Collins and Goodman-Collins 2010).

### Mapping Needs

No combination of existing mapping efforts in the Bay Area provides the map that is needed to protect the ecological services of the T-zone. While some aspects of the needed map are being addressed, there is currently no map of the T-zone Types or the Sub-zones. This minimizes the ability of T-zone designers and managers to assess constraints and opportunities to restore or enhance T-zone services.

**Table 4.3.** Brief descriptions of mapping efforts in the Bay Area relevant to transition zone restoration and protection, with initial considerations for improving the efforts with additional modeling and transition zone data.

Existing Mapping Efforts	Sub-zones			
	SZ1 Evolutionary Adaptation	SZ2 Refuge	SZ3 Flood Control	SZ4 Sea Level Rise Accommodation
Fulfroost and Thomson (in Preparation)	GIS model of the landward limit of backmarsh based on DEM and interpolated tidal datums			
H.T. Harvey and Associates 2002	Combination of field methods and remote sensing used to estimate bayward limit of freshwater discharge effects on tidal marsh vegetation			
Collins and Goodman-Collins 2010, Harvey et al. 1978, NOS 1975	Relative abundance of tidal marsh plant species along field transects used to assess the landward extent of the backmarsh			
SFEI 2013; FEMA 2003			Local Maps of flood infrastructure based on LiDAR plus maps of flood-prone areas used to estimate landward extent of flood control needs and opportunities.	
SFEI 2014	Physical and botanical field indicators used to delineate the upstream and downstream limits of local heads-of-tide.			
NOAA 2012				GIS model used to forecast future Bay margins based on interpolated tidal datums, coarse DEM, and selected sea level rise rates.
Possible Augmentations of Existing Mapping Efforts	SZ1	Use field methods (e.g., NOS 1975, Harvey et al. 1978, H.T. Harvey and Associates 2002, Sawyer et al. 2008) to calibrate remotely sensed spectral and structural signatures of plants indicative of the landward and bayward aspects of SZ1, and the landward extent of SZ2, and add these signatures to a hybrid of existing GIS models (e.g., based on SFEI 2010, NOAA 2012, Fulfroost and Thomson in Preparation) to improve maps of SZ1 and SZ2.		
	SZ2			
	SZ3	Combine existing numerical hydrological models used to manage flood risks with new models of combined effects of storm surge and terrestrial runoff plus new maps of flood infrastructure (SFEI 2013) and HOT (SFEI 2014) to forecast effects of sea level rise and changing precipitation patterns on flood hazards, and to test the efficacy of dispersing riverine floodwaters across accommodation spaces, tidal marsh plains, and diked baylands (i.e., SZ3).		
	SZ4	Augment the existing federal approach to sea level rise mapping (NOAA 2012) with high-resolution DEMs, new flood infrastructure maps (SFEI 2014), BAARI (SFEI 2011), and detailed land use maps to provide local estimates of accommodation space needs and opportunities for selected sea level rise scenarios.		

## **TRANSITION ZONE FUTURE CHANGE**

There is a growing consensus among climate scientists that the Bay Area will undergo significant climate change consisting of a rapidly rising Bay and changes in temperature and precipitation patterns by 2100. Although many factors can affect the local or sub-regional accuracy of the forecasts (e.g., Knowles and Cayan 2004, Flint and Flint 2012), most point to higher average air temperatures, greater warming in summer than winter, earlier warming in the spring, more variable rainfall between years and decades, more intense rainstorms, longer and perhaps more extreme droughts, increased flood risks (due to more intense rainstorms), and decreased average annual stream discharge (Knowles and Cayan 2004, Cloern et al. 2011, PRBO Conservation Science 2011, Cayan et al. 2012). Simply stated, the region can expect increased aridity in the context of more extreme weather events. The estimates of how high the Bay will rise by 2100 range broadly, due mainly to different assumed rates of global greenhouse gas accumulation. For the purposes of BEGHU, it is assumed that the mean level of the Bay will rise 1.4 to 5.5 ft above the current mean level (NRC 2012). The likely response of the tidal marsh plain to these different sea level rise scenarios is discussed in Chapter 2. With regard to the T-zone, the different rates of sea level rise mainly translate into different rates of upstream and landward migration of the T-zone or its compression against the built environment. Additional factors affecting these possible responses are outlined in the following conceptual model of T-zone formation, structure, and services.

### **Conceptual Model of Drivers and Factors of Transition Zone Form, Structure, and Services**

Successful T-zone restoration and management requires understanding how its dimensions, condition, and ecosystem services vary around the Bay due to basic controlling processes and factors. The following conceptual model (Figure 4.7) identifies the major controls on the extent and condition of the T-zone as defined and described above. The purpose of the model is to help identify the controls that can be intentionally managed to restore and protect the T-zone.

According to this model, there are three main drivers of T-zone change: climate, geology, and land use. In this context, geology means the greater formative processes of a landscape, including tectonics, seismicity, and orogeny, as well as its geologic structure. The drivers of T-zone change operate through a complex network of interacting factors.

Of these three drivers, only land use can be managed to affect desired T-zone conditions. For example, waves along the shoreline, groundwater height, and runoff can be managed to achieve the surface hydrology, soil conditions, and topography necessary to achieve target levels of ecosystem services, while accommodating factors that cannot be easily managed such as seismicity, deep landsliding, changes in precipitation, and sea level rise.

The T-zone changes over time. It is subject to relatively gradual changes in some factors, such as sea level rise, biological invasion, subsidence due to groundwater extraction, and increases in recreational use. It can also be subject to sudden and sometimes extreme natural events with major consequences, such as levee breaches and wild fires. The relative importance of the various controlling factors can also change. For example, as sea level rise accelerates it will probably gain importance relative to other factors and processes.

Comprehensive knowledge of the historical T-zone throughout the Estuary would be helpful to calibrate the model and to understand all the restoration possibilities. The existing remnants of the historical T-zone do not represent its full range of natural condition, and they have been severely altered by land use (Shellhammer 1982,

Shellhammer et al. 1982, Goals Project 1999). Furthermore, the construction of levees has added to the possible range of T-zone conditions, and levees are likely to be part of the future T-zone.

### General Considerations

While the relationships among the basic processes and factors that control the extent and condition of the T-zone can be conceptualized (see Figure 4.7), there are few models to quantify the controls or predict the resulting levels of T-zone services. Models for predicting tidal marsh response to sea level rise (see Chapter 2), or to estimate levels of wildlife support, or to estimate the ability of the T-zone to attenuate wave energy pertain to the tidal marsh portions of Sub-zones 1 and 3 (SZ1 and SZ3). Models used to predict flood hazard reduction through flood water storage pertain mainly to Sub-zone 3 (SZ3). There are essentially no quantitative models to predict levels of other services of the T-zone. The following discussion of the likely responses of the different types of T-zone to climate change and sea level rise is therefore general and qualitative.

The T-zone will be affected by climate change in local watersheds as well as sea level rise (see Figure 4.7). The expected increase in the intensity of rainstorms could result in more erosion of hillsides and streams, which in turn could increase the volumes of sediment delivered to the Hillslope-Alluvial Fan T-zone, Bluff T-zone, and Riverine T-zone. The expected increase in dry season air temperatures and the possibility of longer droughts could result in more frequent fires in the undeveloped landward Sub-zones of each T-zone type. It's very difficult to predict how the terrestrial vegetation of the T-zone will be affected by climate change, but increased invasion of non-native plant species is likely, given that these species tend to exploit disturbed environments. Changes in the plant community of the T-zone will in turn lead to changes in how the T-zone supports wildlife.

The basic effects of the rising Bay on T-zone conditions are perhaps more predictable. As the Bay rises, the T-zone will tend to migrate landward, if there is adequate accommodation space. Otherwise it will be increasingly compressed until Sub-zones 1-3 are very narrow or drowned. Since the diversity of services of the T-zone increases with the number of intact Sub-zones, and since the levels of service of any Sub-zone tend to increase with its width, compression of the T-zone will result in a loss of both the diversity and levels of its services. This highlights the importance of a broad SZ4 that can accommodate the landward migration of all the Sub-zones.

While having a broad SZ4 will certainly increase the resilience of the T-zone to climate change, it will not guarantee that the T-zone will remain intact. The landward migration of the T-zone is not a simple process. Migrating plant species may encounter unsuitable soils and moisture regimes, resulting in some amount of secondary succession<sup>4</sup>. As the plant communities of the four Sub-zones change, so will their habitat functions. There may be no way to prevent some degree of net ecological change in the T-zone due to climate change and sea level rise, even if the T-zone has ample accommodation space.

Extreme weather events can significantly affect conditions of the T-zone. For example, the distributions of plants and animals tend to reflect variations in soils, environmental moisture, and disturbance regimes along the elevation gradient from SZ1 through SZ4. These environmental factors are commonly influenced by road grades, levees, tide gates, and other unnatural structures that control either tidal or riverine flooding. Extreme storm events that overtop these structures can suddenly alter conditions of the T-zone, especially for SZ2-4, by changing soil and moisture conditions. An increase in the frequency of riverine flooding has the potential to increase the rate of change in T-zone conditions, causing them to be less predictable and less manageable.

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<sup>4</sup> Ecological succession is the process by which a biological community evolves over time. Primary succession involves the initial colonization of an area by vegetation. Secondary succession involves the replacement of some plant species in an area by others plant species.

Likewise, as the average height of the Bay increases, and the HOTS migrate upstream, the likelihood that wind-generated waves, boat wakes, and extreme high tides, including “king tides”, will overtop levees and berms increases. Conditions in the T-zone are likely to be as affected (or perhaps more affected) by such extreme tidal events than by the increase in average Bay height.

Under natural conditions, the T-zone can be resilient to climate change and extreme weather. For example, barrier beaches can naturally gain height with the deposition of materials during storm wave run-up (Baye, unpublished data), and alluvial fan vegetation buried by episodic riverine flood sedimentation can regenerate after a few years (Goman et al. 2008). The perennial native vegetation of the backmarsh has also shown the ability to grow through flood deposits of sediment (Allison 1996). In general, tidal and non-tidal wetlands show remarkable resilience to moderate levels of disturbance and normal variations in hydrology (e.g., Mitsch and Gosselink 2007, Culbertson 2001). This does not mean that the ecosystem services of the T-zone will withstand climate change and sea level rise without human intervention, but rather that careful intervention is likely to be able to sustain appreciable levels of the services.

The impacts of climate change on the T-zone are likely to accumulate over time. For any given rate of sea level rise and T-zone migration, the frequency and extent of T-zone compression will tend to increase. As MHHW increases, the likelihood of levee failures and major riverine flooding will also increase. If the Bay rises fast enough and long enough to drown the tidal marshes, waves from the Bay that would otherwise be damped by the marshes will have greater energy, and the rate of erosion of SZ1-2 could increase.

If nothing is done to protect and restore the T-zone, its ecosystem services will decline. There will be fewer kinds of service and their levels will be reduced. The primary reasons for this are the lack of accommodation space (SZ4), the increased vulnerability of the T-zone to erosion and disturbance as the adjoining tidal marsh erodes (mostly effecting SZ1-2), the increased vulnerability of the T-zone to biological invasion due to increased frequency and magnitudes of disturbance (SZ-4), and increased fragmentation of the T-zone along the bayshore due to its extreme compression against the built environment. Without intervention to prevent or minimize these impacts, they are likely to become increasingly severe as the climate continues to change and the Bay continues to rise.

### Responses of Transition Zone Types

The impacts of climate change and sea level rise are likely to vary among the types of T-zone. For a more detailed description of the T-zone types see Appendix 4.2.

Artificial Levee. Bayshore levees are typically very narrow, steep, and poorly maintained T-zones that were not designed as habitat or for long-term flood protection. Without intervention, their services as a T-zone are likely to be lost because they are fixed features that cannot dynamically build upward and landward in response to increasing Bay levels. Unlike the natural T-zone types, this artificial type is not resilient to extreme weather events. The levees are at risk of wave erosion and overtopping. The “Horizontal levee” is a recent concept for building resilience and ecosystem services into unnatural bayshore levees (The Bay Institute 2013). According to this concept, the bayshore levee is augmented with carefully graded fill that extends the T-zone bayward to create a wide, low-gradient, terrestrial slope. Non-tidal wetlands can be designed into the Horizontal Levee. The concept is perhaps most applicable to urbanized areas that lack accommodation space. Implementation of the Horizontal Levee concept might require partially filling diked Baylands or filling shallow areas of the Bay adjacent to the existing T-zone.

Bluff or Cliff. This T-zone type is characterized by near-vertical slopes between marshes and uplands. Given its steepness, the bluff-cliff type is very narrow. Its steepness results from either the presence of titled, erosion-resistant bedrock or fractured, friable bedrock that has been undercut by waves. Freshwater seeps are common along fractures

in the bedrock and along the cliff bottoms. This seepage can sometimes help weaken the bluff or cliff and make it more susceptible to erosion.

Since the adjoining tidal marsh tends to reduce the energy of ship wakes and wind-generated waves, the loss of the marsh due to rapid sea level rise could increase the rate at which the waves undermine the bluff or cliff, resulting in mass wasting processes, such as slumping and landsliding. Such erosion would likely cause the bluff or cliff to retreat (i.e., migrate landward), and therefore would not necessarily cause a loss in this T-zone type. One possible consequence of increased erosion of a bluff or cliff is that the eroded material accumulates as an area of natural, supratidal fill over the top of the backmarsh, resulting in a narrow sea level rise accommodation space (i.e., a narrow SZ4).

Valley Plain – Alluvial Fan. Over long time periods, eroded sediments from the steeper areas of local watersheds have accumulated in less steep downstream areas, forming gently sloping alluvial fans and valleys. Broad T-zones form on the fans and in the valleys that extend to the Bay. The landward Sub-zones (SZ3 and SZ4) of this T-zone type have mostly been converted to agriculture and more intensive land uses during the last two centuries. However, there are significant efforts underway to reestablish open spaces on alluvial fans and in the valleys adjoining the Bay, using combinations of natural and engineered topography and hydrology.

Looking forward, the expected increase in rainstorm intensity and riverine flooding could be used to increase the supply of sediment through the fans and valleys and to the backmarsh, pushing the T-zone bayward and enlarging its accommodation space. At the same time, the expected decrease in average annual precipitation and the expected increase in air temperatures during the dry season could decrease surface and groundwater flow through the fans and valleys, and thus reduce or eliminate the associated slope wetlands, depressional wetlands, and brackish marshland that are naturally associated with this T-zone. Creative ways to improve safe yields of sediment and to assure adequate flows of water through the fans and valleys that adjoin the Bay are needed to protect and restore the Valley-Fan T-zone.

The urban development that has occurred along the bases of alluvial fans and gently sloping hillsides, and along the valley bottoms adjoining the Bay will be subject to increased susceptibility of flooding due to climate change and sea level rise. These areas will be the focus of much public debate about whether to build and maintain a flood control infrastructure to protect existing land uses or retreat to safer lands. The outcome is likely to vary around the Bay, depending on the value of the existing land uses. The debate is more likely to favor T-zone restoration if the long-term costs to maintain the infrastructure are considered, given that the Bay is likely to continue rising at a rapid rate for the foreseeable future. In the meantime, the costs to secure the remaining areas of relatively undeveloped accommodation space associated with valleys, plains and alluvial fans will continue to rise.

Barrier Beach. Barrier beaches are narrow, relatively steep, naturally high areas between tidal flats and tidal marshes created by the deposition of materials during wave run-up (Figure 4.8). In profile, they often consist of sand foreshores grading up to steeper mixed coarse sand, gravel, and shell hash. The vegetation on top of barrier beaches typically resembles that of the landward limits of the backmarsh. The barrier Beach T-zone usually only consists of SZ1-2. It persists where there are adequate supplies of sand delivered by wind-generated waves. Barrier beaches can migrate landward as the Bay rises. Where tidal marshes drown, barrier beaches could persist along the landward margin of the backshore.

The supply of sand needed to sustain barrier beaches is mainly provided as bedload by local rivers and streams. Its ability to reach the Bay and thus sustain the barrier beaches is limited by its entrapment behind dams and within constructed flood control channels. Sand is typically a very small portion of the total

sediment load of local streams, except during high flows, at which time nearly 70% of the sediment load can be sand; as much as 50% of the annual load can be sand during very wet years (Patrick et al. 2013). While waves are locally generated by wind, waves propagating across deeper water experience less attenuation and are consequently more erosive. Mudflat erosion due to inadequate supplies of fine sediment (i.e., silts and clays) therefore exacerbates sea level rise by further increasing water depths and hence the erosive power of waves. The interaction between wave erosion and coarse sediment transport differs among geomorphic settings, but Figure 4.9 illustrates how barrier beaches are not sustainable without a supply of coarse sediment. For some “pocket marshes” that exist in coves along cliffs and bluffs, the source of sediment to build barrier beaches can be cliff or bluff erosion.

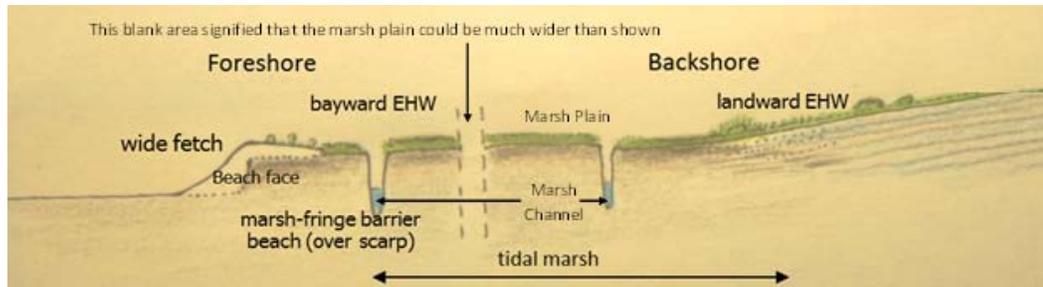
Riverine. Many factors will affect the future conditions of the Riverine T-zone. It is characterized by spatially and temporally complex interactions of fluvial, tidal, and terrestrial processes that create dynamic gradients in salinity, stream power, channel form and structure, and water chemistry, all of which can be affected by climate change. It is important to consider that, under natural conditions, the riverine channel tends to migrate slowly across its valley, with the Sub-zones of the Riverine T-zone generally extending laterally from the riverine channel. Unfortunately, many streams around the Bay are moderately to deeply incised, laterally constrained by land use and riverine levees, or fixed in place as flood control channels, such that they cannot migrate and their Sub-zones are very narrow. Furthermore, the HOTs of urban streams are usually constrained in the upstream direction by artificial grade control structures associated with bridges, trestles, culverts, and sewer lines.

These grade control structures and artificial levees, in combination with the predicted increase in Bay levels and increased frequency of large stream discharges may result in sudden large-scale changes in conditions of the River T-zone. As the Bay rises, the higher tides will eventually overtop the grade control structures, allowing the HOTs to rapidly advance upstream. In these cases, the upstream migration of HOT is therefore likely to proceed in at least two phases, one before the grade-control structure is regularly overtopped at high tide, and one afterward (SFEI 2014).

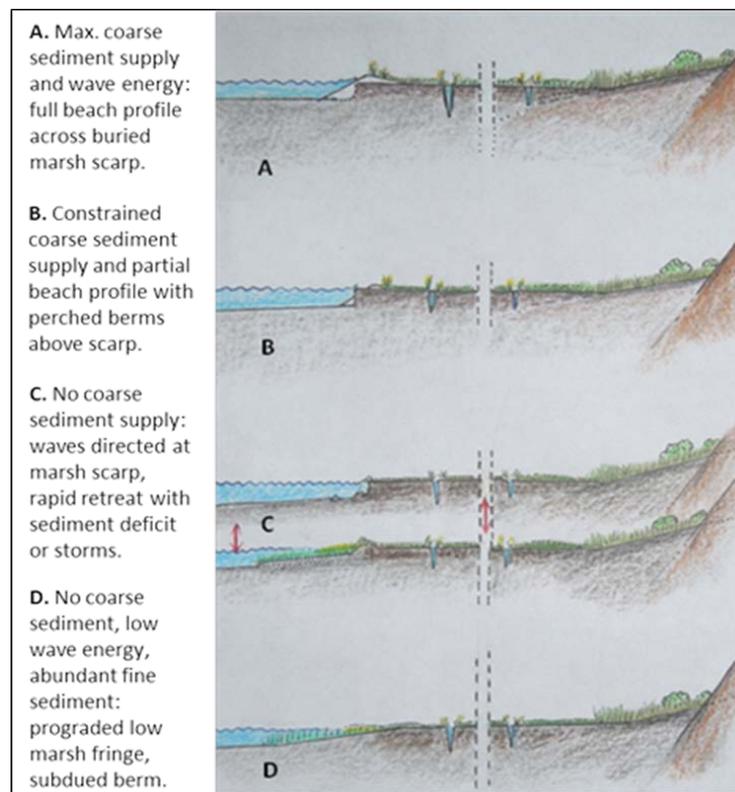
After the obstacles to its upstream migration are overcome, the HOT will suddenly migrate upstream. The rate of migration will be greater where the channel bed is not steep. The upstream HOT migration could significantly increase local risks of riverine flooding during high tide, while the rising Bay increases the risk that wave run-up or storm surges will overtop levees along the tidal reaches of Riverine T-zone.

To prevent this increase in riverine and/or tidal flooding, levees will need to be raised and extended upstream. Much riverine levee work can be anticipated as part of many local responses to increased flood risks. However, alternatives to longer and higher levees should be considered. For example, the restoration or construction of terrestrial floodplains should be considered, as should the ability to shunt floodwaters across tidal marsh plains during low tide and into diked baylands during high tide. In some areas, it might be possible to move riverine levees farther apart, to make room for floodplains between the levees (i.e., inset floodplains of multi-stage channels). Flood control designs can be integrated with the realignment of infrastructure and planned retreat of land uses at the landscape scale to create accommodation spaces with abundant riverine ecosystem services. The concept of incorporating diked baylands and tidal marsh restoration into flood control planning is already gaining acceptance (SFEP 2014). One can imagine a stream with restored floodplains above the HOT, new floodplains created by filling landward areas of adjacent diked baylands, new floodwater retention basins or restored tidal marsh in the bayward areas of the diked baylands, and Horizontal Levees bayward of the retention basins. These concepts and others could be integral elements of landscape designs that reconnect the Bay to its local watersheds in ways that

restore the ecosystem services of the baylands as a whole (SFEP 2014). Implementing such plans will require deciding how to manage the increasing risk of tidal and riverine flooding, including how to repurpose diked baylands. Future policy and decision-making around watershed-based sediment management and flood control will largely determine if flooding is used to nurture the Riverine T-zone. Watershed-based sediment management, as envisioned for rivers and streams impaired by fine sediment (e.g., SWRCB 2007), should consider the effects of sediment management on the Riverine T-zone and other components of the baylands. If a sediment management plan (e.g., AMBAG 2008, SANDAG 2009) is developed for this region, it should fully incorporate the sediment needs of the baylands.



**Figure 4.8.** Profile of the Barrier Beach T-zone. Vertical dashed lines indicate that an unnecessary portion of the profile has been intentionally omitted.



**Figure 4.9.** Cross section of tidal marsh landscape showing effects of sediment supply and wave energy on barrier beach development. Vertical dashed lines indicate where a redundant portion of the profile has been intentionally omitted.

## TRANSITION ZONE DESIGN AND MANAGEMENT

The T-zone presents difficult management challenges because of the need to balance demands for numerous ecosystem services often requiring different management practices with limited resources. Managing natural services, such as wildlife refuge, requires strictly limiting public access to the T-zone, whereas managing the T-zone for various kinds of cultural services requires providing abundant access. The challenge is greatest in the urban environment, where the pressures for access are most intense and the wildlife habitats are most threatened. Also, numerous forms of artificial drainage, including urban runoff and discharges from publically owned treatment plants, connect the urban environment to the Bay through the T-zone. This and other infrastructure, including roadways, fuel lines, and communications and power transmission lines bring additional management interests into the T-zone. Meeting the challenge of managing the T-zone for a diverse set of services requires ongoing coordination between managers and agencies at all levels of government. There is no single “T-zone agency” or advisory group that can provide all the authority or expertise that are needed. However, with the required breadth of technical input, the conflicts among T-zone management objectives can be mitigated through T-zone design.

This purpose of this section is to provide basic guidance to T-zone designers and managers. This guidance is intended to maximize the chance for project success by helping planners and managers choose or develop appropriate design concepts. It is based on the assumption that T-zone restoration is a new subject for many ecologists, engineers, and managers who will be involved in implementing T-zone restoration projects. The guidance is therefore fundamental. Given that there is relatively little experience with the restoration of the various T-zone types around the Bay, prescriptive recommendations are not possible. At this early stage of T-zone restoration science and engineering, pilot projects are needed to test various design concepts and management approaches. Additional management strategies may also be determined as mapping or other methods to delineate T-zones are developed or refined.

### General Principles for Local Transition Zone Design and Management

This set of general principles should be considered at the beginning of any T-zone restoration or enhancement project, regardless of T-zone type.

Emphasize Collaboration and Partnership. All designs should be developed with an expert understanding of the permits needed to implement the designs, with input from all agencies responsible for any aspect of the T-zone, and with the ongoing advice and review of special interest groups and the public. To the degree possible, projects should be co-sponsored by multiple agencies.

Identify the Target T-zone Type(s). The first technical step in T-zone planning or restoration is to decide what type of T-zone is best suited to the restoration site. Most projects will only involve a single T-zone type. However, large projects might involve multiple types. Historical Ecology (HE) can guide the selection of the target T-zone type by revealing what type existed historically at or near the project site and might therefore still be suitable. The HE analysis can also reveal the site-specific form of the T-zone, including how and why the Sub-zones varied in width. The goal of HE evaluations is not necessarily to recreate the historical landscape but rather to reveal the site-specific T-zone types and their particular characteristics that could still be compatible with local controlling factors. HE can be invaluable for fully understanding the local restoration opportunities.

Identify and Assess Dominant Local Physical and Biological Processes and Needed Expertise. To the full extent possible, T-zone designs should accommodate the local physical and biological processes and

extreme events that are likely to control the target levels of selected T-zone services. Review the factors and processes controlling the formation and condition of different T-zone types (Figure 4.7 and accompanying text). Consider the likely effects of climate change on these factors and processes. While there are numerous processes acting on any T-zone, its form, structure, and services are usually controlled by a few dominant processes. When assembling a project design team, identify the particular scientific and engineering expertise required, based on the assessment of what natural or anthropogenic processes and events control the form, structure, and services of the site, now and into the future. For example, the design of a Levee T-zone that is likely to be at risk of wind-wave erosion should involve experts in coastal engineering, geomorphology, and ecology who can explore natural and artificial solutions to dissipate the wave energy. For the design of the Riverine T-zone, information about storm and annual hydrographs plus sediment loads is always important. The design team should therefore include a fluvial geomorphologist and hydrologist. Biological invasion by non-native plant species is a process of special concern for most restoration projects that requires the expertise of a botanist and/or restoration ecologist. T-zone projects typically require careful determination of both tidal elevations and geodetic elevations, and should therefore involve experts in these technical subjects.

Evaluate Local Landscape Constraints. Many T-zone designs must accommodate existing infrastructure and land uses that constrain the kinds or levels of ecosystem services that the T-zone can support. The design team must take into account any and all infrastructure above and below ground, plus rights-of-way, needs for emergency access, and the kinds and levels of existing recreational use. Ways to minimize constraints and conflicts through project design and timing should be explored. For example, the T-zone has conventionally been regarded as a suitable location for water lines, sewer lines, gas pipe lines and high-tension electrical power transmission lines. Much of this infrastructure obstructs any modification of the T-zone. The managers of this infrastructure typically charge restoration projects for its protection or realignment, which can greatly increase project costs. However, the costs can be reduced if the projects are planned to coincide with the replacement or repair of such infrastructure.

Mosquito control can be an essential and sometimes expensive aspect of T-zone management. Mosquito Abatement Districts typically advise project planners to consider natural processes of source control (e.g., wave action, tidal flushing, and predation), as well as water management and biological control as needed. The Districts may need to be compensated for ongoing mosquito control efforts. Public outreach and education will be needed to manage nuisance levels of mosquito production and associated health risks.

Identify the Target Kinds and Levels of T-zone Services. The steps outlined above provide the basis for exploring the kinds and levels of services that the T-zone should provide. Review the discussion on ecosystem services and their relationship to T-zone types and Sub-zones. Consider any local requirements to implement water quality control plans, habitat conservation plans, and recreation plans. The focus should be on what is needed, rather than what is simply desirable. Vet the decisions about services with the affected communities of people.

Select the set of services that is most suitable for the target T-zone type. Focus on achieving the few services that are most likely. Be conservative; while it is certainly possible that many services will be supported by the project, it is only likely to provide sustainably high levels for a few services. It will not usually be possible to quantify the target levels of all selected services. In fact, climate change increases the uncertainty of such quantifications. In general, physical services such as sea level rise accommodation, flood water storage, and erosion control are more easily quantified than biological or ecological services,

such as buffering and wildlife support. The selected levels and kinds of service must be consistent with the targeted type of T-zone, reference conditions, and the landscape constraints of the project site.

Identify and evaluate reference sites. These are examples of the type of T-zone selected for the project that exhibit the best achievable or least altered conditions. To the degree possible, the project should be designed to achieve the levels of selected ecosystem services evident among the reference sites, adjusted for inmitigable landscape constraints. For some T-zone types, reference condition may no longer exist. In these cases, the reference conditions should be inferred from historical evidence.

Design the Sub-zones and Refine the Target Kinds and Levels of Service. At this stage of the project design, it should be possible to sketch the T-zone in plan-view and profile, such that the approximate widths, topography, elevations, and vegetation of each Sub-zone can be visualized. Engage appropriate experts in ecology, hydrology, and tidal elevation reckoning to review the drawings. Adjust the target kinds and levels of services as warranted based on the conceptual Sub-zone designs. Be realistic; do not plan to extract more kinds or levels of service than the site can realistically sustain. However, consider possible future expansion of the project, and aim for designs that do not diminish the possibility for such expansions.

Build a Conceptual Model. Use the results of the Historical Ecology analysis, the evaluation of site-specific controlling processes, and the evaluation of reference conditions to build a conceptual model of the likely cause-and-effect relationships among the controlling processes and the selected kinds and levels of ecosystem services. Explicitly identify basic assumptions that underlie the causal relationships. Use the model to identify key constraints and design attributes that are most likely to affect project success. The model should serve as a framework for estimating project timetables, costs, and risks. Update the model with new information as the project design evolves. Completion of the conceptual model can result in further modifications of the conceptual design of the project and the target services.

Engage the Community. Public outreach and education are vital to increasing the public awareness of the importance of the T-zone and to create the political will to support restoration and protection of the T-zone into the future.

Consider Relationships to Other Projects. This is a major step in the T-zone design process. Each new project should be considered in the context of other existing or intended projects. Of special concern are any other projects in the same landscape, meaning that they are likely to influence the levels and kinds of ecosystem services they can sustain. A practical initial approach to this analysis is to review whether the projects are likely to influence each other's hydrology, sedimentation, vegetation, or wildlife support. A proactive approach is to develop a conceptual plan of T-zone restoration for the entire landscape of the project. This is the best way to assure that all the projects that might eventually occur in the landscape work together to achieve the highest levels of the most diverse array of ecosystem services possible. This is also a way to develop long-term plans for minimizing landscape constraints.

Think ahead. Existing management practices might be modified to prepare future areas targeted for T-Zone restoration.

In agricultural lands, consider allowing the formation of farmed wetlands, creating ponded areas along the inboard toes of levees and at sumps, encouraging tall vegetation as wildlife habitat along fences, delaying spring harvests of hay and similar crops to protect breeding wildlife, using rotational grazing practices to

encourage habitat diversity, providing livestock with water away from riparian areas, and fencing livestock from seasonal wetlands during wet periods.

In urban areas, consider establishing setbacks along watercourses that link tidal marshes to healthy riparian corridors, and disallowing fences (or requiring fences to be elevated off the ground) to enable wildlife movement through the built environment.

The landscape approach to T-zone design might involve realignment of existing elements of the project site. For example, consider locating heavy-use recreational trails at inland locations or at the landward edge of SZ3, such that human access is restricted from bayward areas of the T-zone that support sensitive wildlife.

It is important to understand the soil characteristics and geotechnical properties of imported materials used to create levees, berms, and other T-zone features. Several restoration projects in the past failed to consider the nature of imported materials or how their condition would change over time. In some cases the materials were too acidic or nutrient-poor to support healthy vegetation. In other cases, the materials lacked the correct geotechnical properties to withstand existing wave erosion.

Design for complexity. For example, consider creating topographic variability at multiple scales. At the large scale, consider grading multiple topographic benches or plains at different heights parallel to the backmarsh or riverine channel that together comprise a stepped gradient in the frequency and duration of tidal or riverine flooding. For small-scale variability, provide swales, hummocks, large woody debris, etc. The occurrence of large woody debris can be very beneficial to many species of wildlife. Designs that are too uniform in terms of their elevations and topographic relief will result in a T-zone that is ecologically simple and therefore less resilient than a more naturally complex T-zone. This guideline is often difficult to implement because most engineering and construction firms are not used to building complexity into their projects. Therefore, consider building projects on a time-and-materials basis with a qualified geomorphologist and ecologist in the field directing the contractor on the details of habitat construction.

Determine How to Measure Progress and Determine Success. A monitoring plan should focus on the minimum data necessary to maintain compliance with all permits and to assess the project's ecosystems services relative to their target levels. The monitoring should focus on what is happening and not why. Understanding the causes of the conditions observed is not necessary unless the conditions are unacceptable. Consider progress as the occurrence of conditions that are consistent with a trajectory of success, defined as achieving the target levels of service. Regularly report the status of the project to the public.

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## **SUMMARY**

The objectives of this chapter are to define and describe the estuarine-terrestrial transition zone of San Francisco Bay, hereafter termed the "T-zone", and to provide science-based guidance for conserving its ecological services in the context of climate change and especially sea level rise.

Efforts to address the ecological and economic threats imposed by sea level rise have begun to focus on the T-zone. There is a growing awareness that its design and management can help mitigate these threats. The T-zone can provide space for the Bay to expand without creating unacceptable flood hazards and without completely

losing the ecological services of the baylands. Many historical cultural resources are associated with the T-zone, and it provides important recreational opportunities, while providing critical habitat for wildlife.

The T-zone is defined as:

*the area of existing and predicted future interactions among tidal and terrestrial or fluvial processes that result in mosaics of habitat types, assemblages of plant and animal species, and sets of ecosystem services that are distinct from those of adjoining estuarine, riverine, or terrestrial ecosystems.*

The T-zone does not have a fixed width. It varies in width from place-to-place and over time. In the landward direction, the width of the T-zone is affected by the vertical range of the tide, the slope of the land, and the locations of built structures that control the upstream or landward movement of tidal water. In the bayward direction, the width of the T-zone depends on the volume of terrestrial runoff entering the baylands. The T-zone only includes the areas of intertidal vegetation that are measurably influenced by terrestrial runoff and other freshwater discharges. It only includes diked baylands that serve to store terrestrial flood waters or that represent future space to accommodate sea level rise.

The width of the T-zone also varies based on the kinds of ecosystem services it provides. For example, a broader T-zone is needed to provide refuge from high tide for marsh wildlife than if such refuge is not provided, and a broader T-zone is needed to accommodate sea level rise for the next century than for the next half-century.

The functional relationship between the T-zone and local watersheds should be emphasized. Most of the inorganic sediment that accounts for the formation and persistence of the tidal marshes is derived from local watersheds. The freshwater runoff from local watersheds creates salinity gradients through the baylands that greatly increase the overall biodiversity of the region. Many wildlife species, including birds of prey and salmon, move between the Bay and local watersheds through the baylands. The Bay and its local watersheds are linked together by the baylands, and the mechanisms of this linkage are the workings of the T-zone.

Seven T-zone types have been identified based on their formative processes. In aggregate, the seven types represent the full range of historical and existing T-zone conditions for the Bay. Each type of T-zone consists of 2-4 sub-zones that provide different suite of services. The kinds and levels of service provided by the T-zone can be controlled to some degree through design and management of the sub-zones.

At this time, there is no regional map of the T-zone as defined here. A regional T-zone map is needed to identify and track restoration opportunities, to assess the relative effects of restoration and ambient climate change, and to evaluate the efficacy of state and federal policies for protecting the T-zone. Local maps are needed to inform restoration design. The optimal mapping approach will probably involve estimating the extent of each type of T-zone and the width of their sub-zones, such that the map can inform the restoration and management of specific ecosystem services.

There are three main drivers of T-zone change: climate, geology, and land use. They operate through a complex network of interacting factors. Of these three drivers, only land use can be managed to achieve target kinds and levels of T-zone services. In this regard, land use includes any activity by people that alters the topography or elevation of the land, or that affects the abundance, quality, and distribution of surface waters.

If nothing is done to protect and restore the T-zone, its ecosystem services will decline. There will be a lack of space to accommodate sea level rise, more shoreline erosion due to waves and ship wakes, increased biological

invasion of the bayshore due to increased disturbance, increased risk of river and creek flooding due to upstream migration of high tides, and increased fragmentation of the T-zone ecosystem due to its compression against the built environment. Without intervention to prevent or minimize these impacts, they are likely to become increasingly severe as the climate continues to change and the Bay continues to rise.

The T-zone presents difficult management challenges because of the need to balance demands for ecosystem services requiring different management practices with limited resources. Meeting the challenges requires ongoing coordination among agencies at all levels of government. However, with abundant technical support, the conflicts among T-zone management objectives can be mitigated through T-zone design. At this early stage of T-zone restoration science and engineering, pilot projects are needed to test various design concepts. In general, each restoration project should engage the public in a process to set ecosystem service goals for the t-zone type that best fits the restoration site, based on an operational understanding of the formative processes, local constraints, and future opportunities for further restoration. T-zone restoration projects should be planned in a landscape or sub-regional context, such that positive synergies among the projects can be maximized.

Methods to assess the existing and restored T-zone should be standardized, such that projects can be compared to each other and to background or ambient conditions over time. Information about the location and status of T-zone restoration projects should be readily available online, and the overall condition and prognosis of the T-zone throughout the region should be regularly explained to the public.

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# Science Foundation Chapter 4

## Appendix 4.1

# Recommended Definition of the Estuarine-Terrestrial Transition Zone

## Guiding Principles and Criteria

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### BACKGROUND

#### Guiding Principles

The Transition zone (T-zone) workgroup developed the following principles to guide its work. These are based on the guiding principles provided by the original Baylands Habitat Goals Report (1999).

- Present a vision of T-zone changes that are needed to conserve or improve the ecological services of the Bay's T-zone. The vision should:
  - Inform local and regional plans to restore the extent and quality of the transition zone without trying to “reach the past;”
  - Support essential physical, ecological, and social services of the T-zone;
  - Give priority to the support of native species of plants and animals, especially threatened, endangered, and other special-status species;
  - Recognize the habitat values of existing land uses such as farming and ranching;
  - Be based on scientific understanding of historical and existing conditions, and sound professional judgment, without undue regard for costs or political conflicts.
- Anticipate the need to modify the vision in the future to reflect improved scientific understanding and practical experience in T-zone restoration and management.

#### Criteria for the Definition

The scientific literature contains many different descriptions and definitions of the area of land and water herein referred to as the Estuarine-Terrestrial Transition Zone (T-zone). Most are based on ecological concepts relating to the distribution and abundance of plants or animals along physical gradients that extend between intertidal and terrestrial or fluvial environments. The terminology for these concepts varies among the various definitions. What is referred to here as the T-zone overlaps with other concepts pertaining to this transitional area.

The workgroup decided that the various local and regional efforts to plan and protect the Bay's T-zone would benefit from a common, standard T-zone definition. It also decided that the definition could be general and did not need to resolve the different uses of scientific terms, as evident in the literature. Based on these beginning agreements, the workgroup developed criteria to evaluate candidate definitions and to finalize the recommended definition, as needed to support T-zone planning and management. The workgroup also recognized the need to align the definition of the T-zone with its needed ecosystem services. The following criteria were developed by the workgroup to recommend a T-zone definition based largely on these reviews and on the workgroup's discussion of the ecosystem services of the T-zone.

1. Dimensionality. The definition should reference the dynamic nature of the T-zone along tidal, fluvial, and terrestrial gradients that vary through space and over time.
2. Inclusivity. The definition should reference the particular kinds of interactions among tidal, fluvial, and terrestrial processes, the functions and services of these processes, and their values to society that distinguish the T-zone from other aspects or components of the greater Estuary, without restrictions in spatial or temporal scale.
3. Credibility. The definition should reflect the state of scientific understanding.
4. Map ability. The definition should either reference landscape indicators of the T-zone or landscape processes that are manifested as remotely visible indicators of the T-zone, such that its regional and local extent can be estimated based on commonly available data.
5. Delineable. The definition should reference physical, chemical, or biological indicators that can be used to determine the spatial limits of the T-zone in the field.
6. Empiricism. The definition should reflect the consensus opinions of the members of the T-zone workgroup based on their collective empirical understanding of the forms and services of the T-zone in the San Francisco Estuary.
7. Parsimony. The definition should be as simple as possible without sacrificing essential meaning for simplicity; the preferred definition should require the least amount of assumption; and the definition should be comprehensive, clear, and not unnecessarily complicated.

The workgroup constructed a definition to meet the criteria listed above. The recommended definition meets some of the criteria better than others, however. It references the suite of intrinsic environmental processes and their services (criterion 1); it is applicable throughout the Estuary and will retain its applicability over time (criterion 2); it is consistent with the scientific understanding synthesized from relevant literature (criterion 3); it is based on the collective experiential understanding of the workgroup members (criterion 6), and it is reasonably simple (criterion 7). Given the natural diversity of T-zone types, the workgroup was not able to develop a definition that meets the mapping and delineation criteria (criteria 5 and 6) as well as the other criteria. The recommended definition is consistent with efforts to estimate the approximate extent of the T-zone based on the current general level of understanding about the controlling physical processes. But, the indicators needed for remote or field-based delineation of the thresholds of

these processes that exactly correspond to the local physical boundaries of the T-zone are not as well known. Some important clarifications of the meaning of the recommended definition are provided below.

- The recommended definition distinguishes the T-zone from the Estuary as a whole, and from the fluvial, terrestrial, estuarine ecosystems bayward and landward of the T-zone.
- With regard to physical processes, the definition is generally met at any given moment by the area that is bounded by the landward influences of estuarine waters on the heights and physio-chemical characteristics of riverine waters; the landward influences of estuarine waters on groundwater height and the physio-chemical characteristics of terrestrial soils and groundwater; and the bayward influences of terrestrial runoff and groundwater on tidal heights and the physio-chemical characteristics of estuarine waters and intertidal sediments.
- With regard to ecological processes, the definition is generally consistent with the spatial distribution of local populations of plants and animals that measurably depend on the T-zone or that are measurably affected by it during some part of their natural history.
- Extreme events, such as “king tides” and major riverine floods strongly influence the range in form, structure, and ecosystem services of the T-zone through space and over time.
- The T-zone surrounds the Estuary, including the tidal reaches of its confluent rivers and streams, regardless of their degree of naturalness.
- The T-zone varies in width along its length and over time due to differences or changes in land use, geology (including topography, hydrology, and soils), and climate (including weather).
- Land use can constrain or release the controlling environmental processes and events, and thus artificially narrow or broaden the T-zone.
- As sea level rises, the T-zone tends to migrate upstream and landward; it is the leading edge of estuarine transgression due to sea level rise.
- Any depiction of the boundary of the T-zone is subject to change.

# Science Foundation Chapter 4

## Appendix 4.2

### Profiles of the Transition Zone Types

#### BACKGROUND

This appendix is designed to serve as a quick reference to the basic characteristics of each of the seven types of the T-zone.

#### Type Hillslope or Alluvial Fan and Valley or Plain

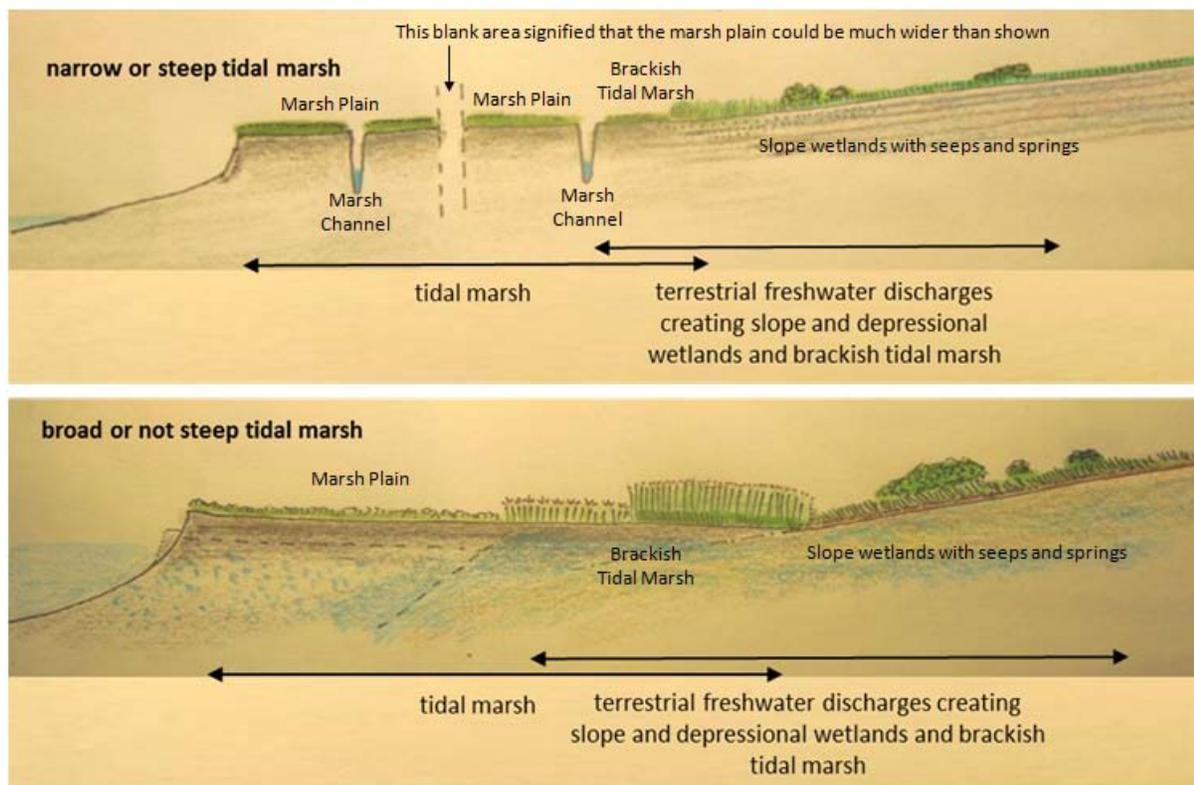


Figure 4.2.1. T-zone associated with hillslope, alluvial fan, valley or plain

#### *Narrative Description of Natural Physical Condition and Setting*

**Landscape Position:** bayward margin of hillslope, alluvial fan, valley or plain.

**General Description.** This type of T-zone exists where tidal marsh borders gently sloping terrestrial lands away from a river or stream. The lands might be the bayward margin of a hillside, alluvial fan, valley or plain. Areas more than 0.25 miles wide corresponding to Sub-zone 1 are described by Cooper (1926) for the historical bayward reach of Santa Clara Valley. The landward part of this type (Sub-zone 4) is often characterized by slope wetlands and depressional wetlands, with seeps and springs in some places, due to

near-surface or emergent groundwater. The backshore portion of this T-zone type is characterized by brackish or freshwater tidal marsh. Where alkali grasslands (grasslands dominated by salt tolerant species) border tidal marsh (e.g. Suisun Marsh; the historical T-zone area near present-day Milpitas), landward and bayward boundaries of the t-zone may be poorly defined by vegetation. If the tidal marsh portion is mature, it can include elongate pans along the backshore.

**Substrate.** Either entisols (recent sediment, no soil profile development) or soil profiles well-developed (including organic-rich sediments created by non-tidal wetlands) contiguous with the landward edge of tidal marsh. Fine textured soils and sediments (mostly fine sand, silt, clay, local coarse channel deposits).

**Slope.** Gently to very gently sloped, near 0% (alluvial flats, valley floors) to 5% (some hillslopes and uplifted alluvial fans).

**Hydrology.** Naturally shallow fluctuating seasonal groundwater (historically drained for agriculture and urban/industrial development), seasonal flooding or overland flow; historical channel avulsion of braided alluvial fan distributaries; extensive groundwater emergence near sea level. Freshwater discharge and terrigenous (flood discharge) sediment transport extends terrestrial influence into tidal marsh bayward of MHHW boundary (visible as contrasting brackish marsh vegetation zone in salt marsh, or fresh-brackish marsh vegetation zone in brackish marsh).

#### Terrestrial vegetation of Sub-zones 3 and 4.

- wet meadow (lowland grassland, valley grassland, sedge meadow, sedge-rush meadow, seasonal marsh).
- alkali meadow (lowland alkali grassland).
- vernal pool grassland phase (depressional wetland pool and swale, annual/spring ephemeral forbs).
- riparian scrub (willow grove or sausal; mixed riparian scrub).
- freshwater slope wetland and depressional wetland (rushes, sedges, willows).

**Regional distribution and abundance.** Historically widespread as the dominant T-zone type of East Bay, South Bay, and much of Suisun and northern San Pablo Bay (Petaluma, Napa, Schellville, Fairfield vicinity). Low slopes and fertile soils favored rapid early agricultural conversion and subsequent urban development. Remnants occur in small valleys of North Bay (often fragmented by diked baylands). ranch lands, open space preserves, and wildlife refuges.

**Type Localities.** Public access lands: (1) Rush Ranch (Suisun Marsh), (2) Sears Point (Sonoma Marshes), (3) China Camp (Marin).

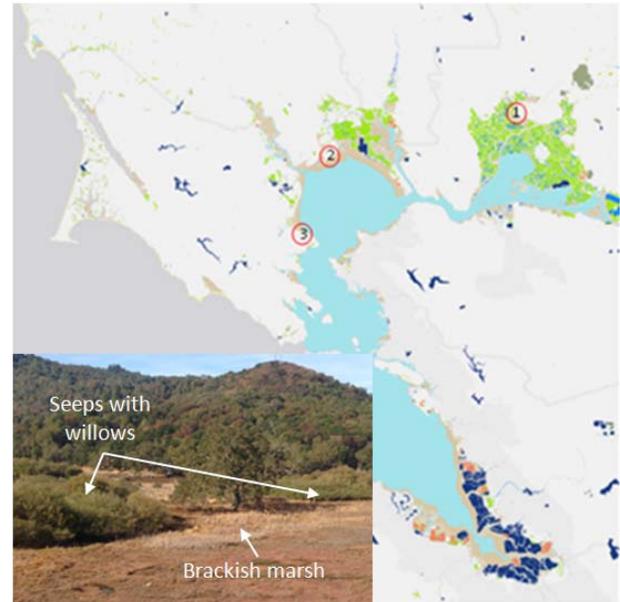


Figure 4.2.2. Type localities for hillside T-zone.

Type: Bluff or Cliff

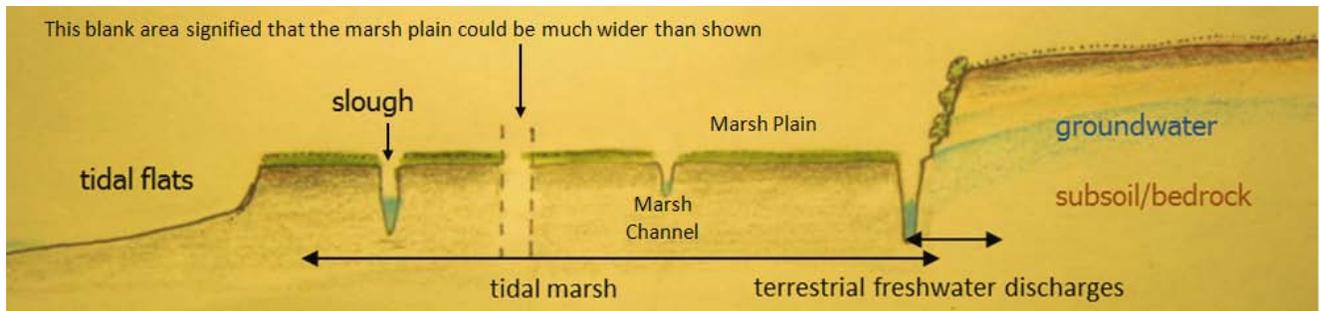


Figure 4.2.3. T-zone associated with bluff or cliff.

*Narrative Description of Natural Physical Condition and Setting*

**Landscape position.** Bayward margin of cliff or steep bluff subject to erosion, sometimes involving a relic slough.

**General description.** This is a relatively rare type of T-zone except on islands and along the hilly Marin County bayshore. It is extremely narrow with an abrupt landward discontinuity in vegetation type due to near-vertical terrestrial slopes of sub-zones 3 and 4. Bluffs and cliffs are generated by erosional processes (including wave-cut bench and channel bank erosion) that are usually constrained by tidal marsh, which can include relict features of channel migration co-existing with prograding tidal marsh.

**Substrate.** Exposure of subsoil, weakly consolidated or cohesive sediments (bluffs), or moderately to strongly resistant bedrock (cliff profiles), or slope debris (talus, slump) to tidal marsh edge.

**Slopes.** Depending on shear strength of substratum, slopes can be steeper than angle of repose of dry sand. Bedrock may be exposed to form near-vertical slopes.

**Hydrology.** Seeps or springs localized in contacts between impermeable and permeable substrates; mostly xeric to mesic, depending on estuary position (fog gradient of Central Bay to Carquinez Strait– most mesic; arid inland)

Terrestrial vegetation of Sub-zones 3 and 4.

- coastal bluff scrub (Central and South Bay, south-aspect slopes);
- riparian scrub (North Bay and Suisun, north-aspect slopes)

**Type Localities.** Public access lands: (1) China camp Islands; (2) Carquinez Strait; (3) south Sausalito.



Figure 4.2.4. Type localities for bluff T-zone.

Type: Barrier Beach

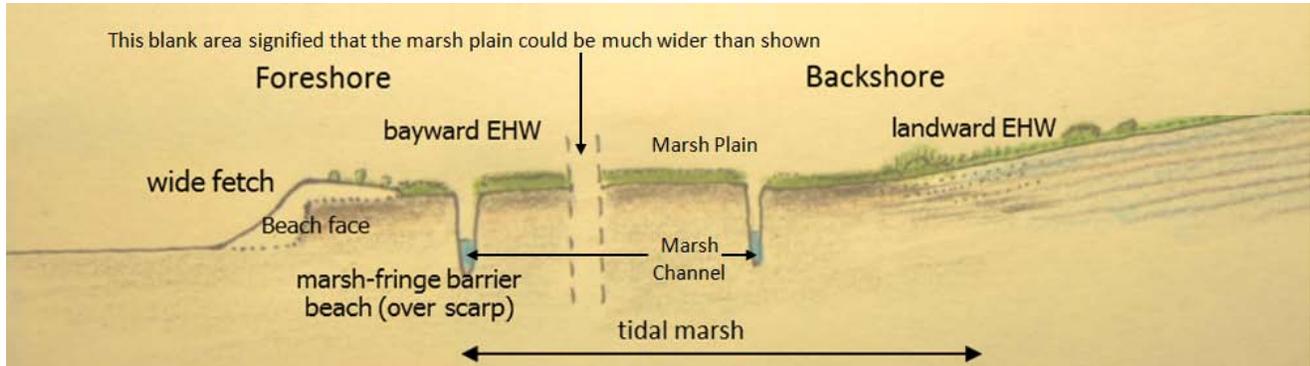


Figure 4.2.4. Barrier Beach

*Narrative Description of Natural Physical Condition and Setting*

**Landscape position:** High wave energy bayward edge of existing tidal marsh at end of long fetches bordering tidal flats; extending as barrier spits across tidal flats or deltas.

**General Description:** Historically prevalent West Berkeley to Alameda (sand), Richmond and Point Pinole vicinity (sand), northern SF Peninsula bayshore (sand, shell) to Palo Alto; Richardson Bay; minor elsewhere. Atwater (1979) attributed natural (San Lorenzo) historic natural salt ponds to marsh berms (barrier beaches).

**Substrate:** Mostly coarse clastic sediments: sands (originating from fluvial, headland bluff, bay sand shoals, Pleio-Pleistocene marine, dune and lagoon or beach deposits of Colma, Merritt formation sources), gravels (headland bluff and cliff erosion); and native fossil oyster shell hash (erosion of shell-rich muds, oyster shell hash bar migration onshore). Grades landward into higher elevation berms composed of mixed fine and coarse sediment and organic debris as coarse sediment supply declines, but subject to material deposition by wind-driven waves. Variations in barrier beaches are largely due to difference in substrate as affected by its sources and wave-energy regimes.

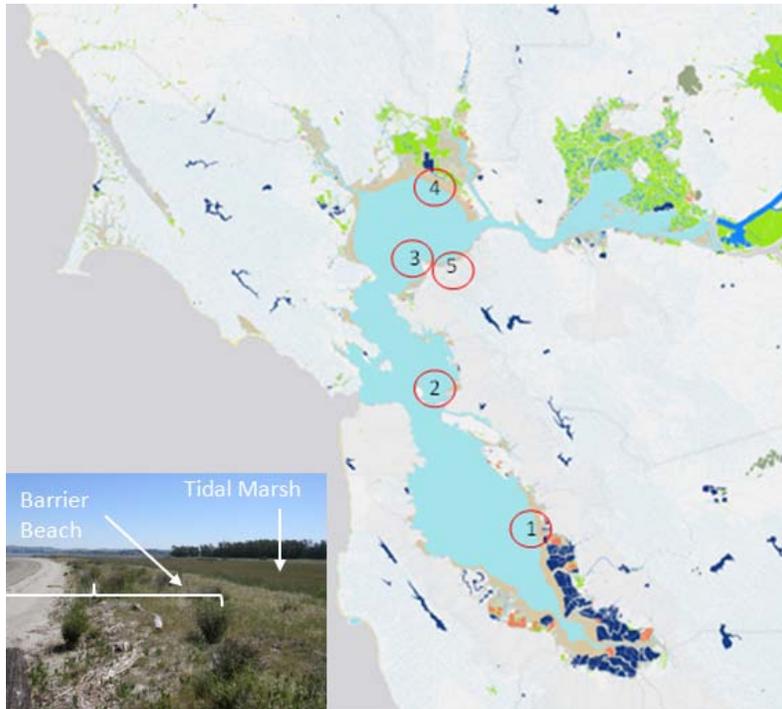
**Slopes:** Steep beachface slopes, especially shell and gravel; gentle slopes of berms, wash-over fans, low aeolian dunes. Elevations range from wave run-up approximately 2 ft above still-water extreme high tides, or more than 2 m where sand dunes form.

**Hydrology:** Shallow freshwater lens perched just below marsh plain. Naturally shallow fluctuating seasonal groundwater. Marsh groundwater discharge through permeable lower beachface. Beaches are deposited by wind-driven waves where coarse sediment supply occurs; limited cross-shore landward transport of coarse sediment (low backwash energy at lower tide) or alongshore (oblique wave approach).

*Terrestrial vegetation types at tidal marsh edge:*

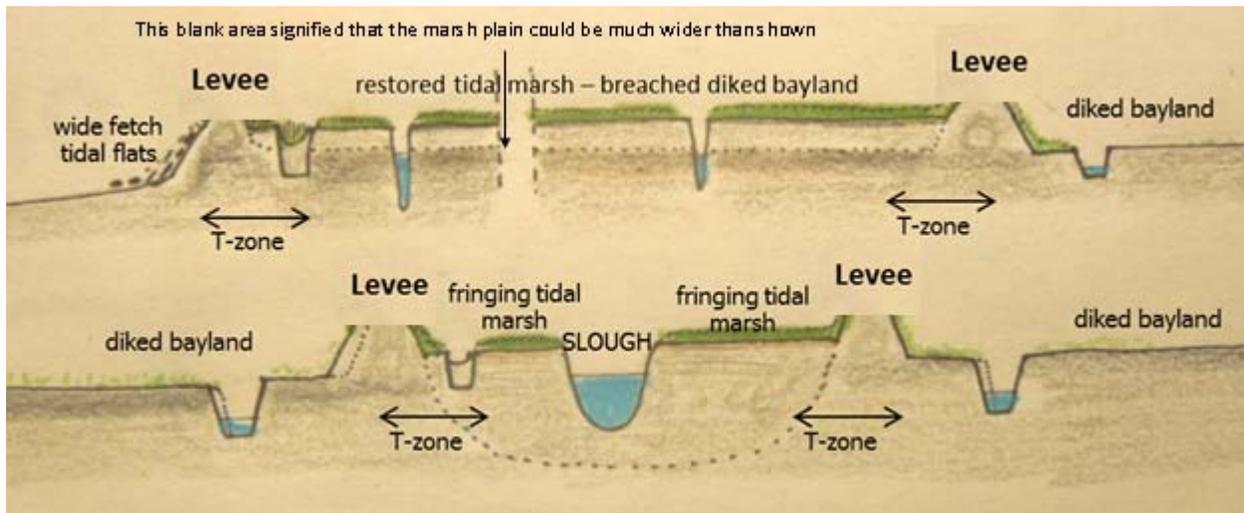
- modern low-energy beach and foreshore pioneers: *Ambrosia chamissonis*, *A. psilostachya*, *Distichlis spicata*, *Cakile maritima*, *Iva axillaris*, *Lepidium latifolium*, *Carpobrotus spp.*
- historical native beach and foreshore vegetation (SF, Alameda, Berkeley, Oakland).
- stabilized beach ridge - high salt marsh succession (*Grindelia*, *Salicornia*, *Frankenia*).

*Type Localities.* Public access lands: (1) San Leandro (Long Beach, Roberts Landing), (2) Radio Beach (Emeryville/Oakland), (3) Point Pinole (Whittell Marsh; sole historic marsh-fringing barrier beach, (4) Highway 37 fringing marsh, (5) Pinole Creek pocket marsh.



**Figure 4.2.5.** Type localities of Barrier Beach

### Artificial Levee



**Figure 4.2.6.** Artificial Levee

#### *Narrative Description of Natural Physical Condition and Setting*

**Landscape position.** Multiple. Bayward edge of tidal marsh (bayfront dike/levee) borders of tidal slough and diked baylands, landward edge of tidal marsh (flood control dike).

**General description.** Extremely narrow zone on outboard levee slope: disturbed by levee capping (maintenance cycle 3+ yr), 1-2 m wide typical. Non-native annual forbs and grasses are frequently dominant over extensive areas. Pervasive throughout estuary: dominant terrestrial transition zone type by linear extent, exceeding natural length of tidal marsh/terrestrial edge due to perimeter dike alignment along historic bay and slough edges of tidal marsh.

**Substrate.** Bay mud dredged from borrow ditches; less commonly composed of imported fill. In Suisun Marsh, peat or muck soils are used to maintain old non-engineered dikes. Bayfront dikes directly exposed to high wave energy of open tidal are often armored with boulder (rip-rap) or concrete rubble.

**Slopes.** Steep 2:1 slopes typical; rarely 3:1 – 5:1 or 7: Dike crests subside and erode, maintained for 1-2 ft freeboard above EHW, but are overtopped by extreme high tides or wave runup as they subside.

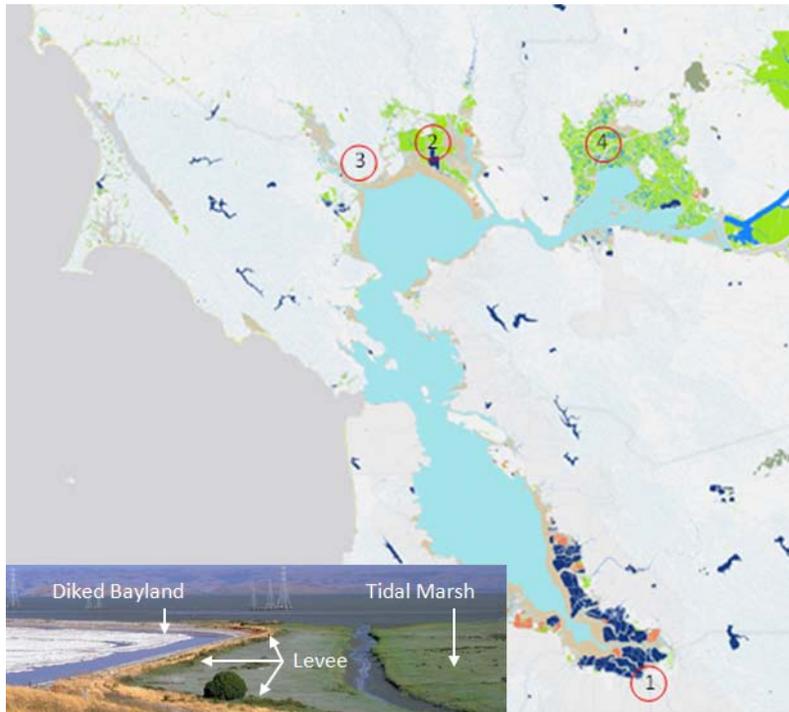
**Hydrology.** Shallow freshwater lens may support salt-intolerant plants: coast live oaks grow on 30+ yr old stable dikes surrounded by salt water in Marin (Richardson Bay and Bahia). Dredged bay mud capping raises soil salinity temporarily, as does overtopping. No significant freshwater seepage occurs through clay dikes detached from supratidal uplands separated by subsided diked baylands.

#### Terrestrial vegetation types at tidal marsh edge.

- high salt and brackish marsh (lower outboard levee slope)
- ruderal forbs and annual grassland (upper outboard levee slope).
- artificial plantings (ornamental, amenity plantings, stabilization plantings, habitat plantings, mixed purpose plantings).

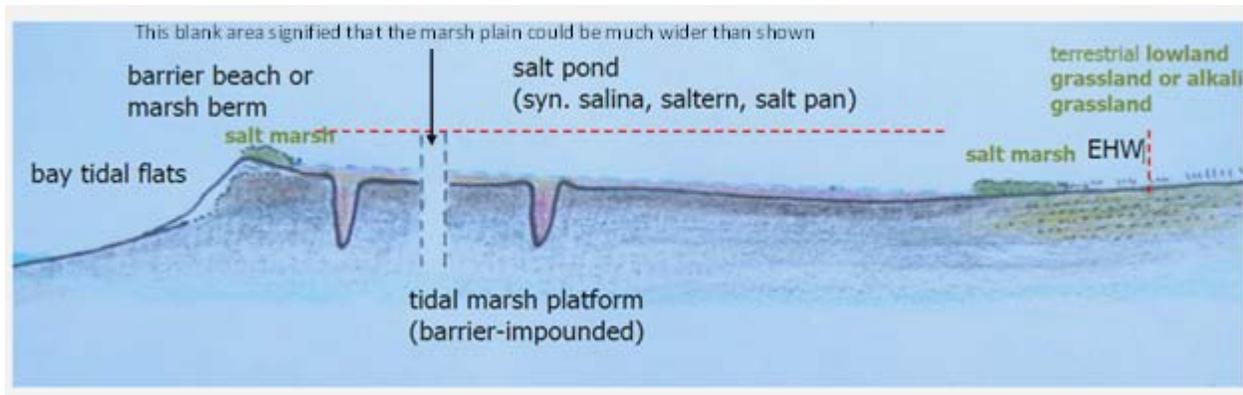
- interior slopes: atypical salt marsh assemblages (high frequency ruderal halophytes).

*Type Localities.* (1) South Bay salt ponds; (2) Napa salt ponds; (3) farmed North Bay diked baylands; (4) Suisun Marsh duck clubs.



**Figure 4.2.7.** Type Localities – Artificial Levee

## Natural Salt Pond



**Figure 4.2.8.** Natural salt pond.

### *Narrative Description of Natural Physical Condition and Setting*

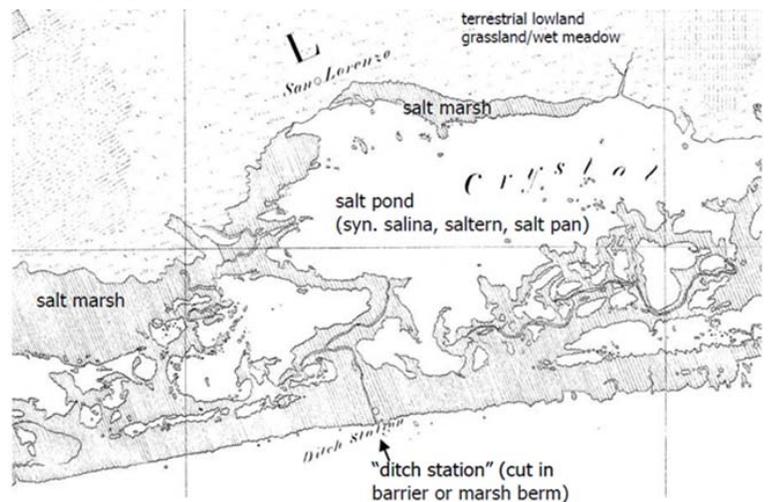
**Landscape position.** Across landward edge of tidal marsh and contiguous alluvial fan below brine flooding elevation range and beach-impounded tidal marsh platform with channel outlets choked or obstructed. (Atwater et al. 1979). Natural salt ponds create gaps (discontinuities) between terrestrial and estuarine emergent vascular vegetation types.

**General description.** Distinctively large and shallow (maximum depth <1m) intertidal ponds of tidal waters trapped behind barrier beaches or marsh berms. Subject to desiccation by evaporation entirely or in part during late summer or fall, causing marine salts to precipitate at edges and sometimes across the pond bottom.

**Substrate.** Halite or gypsum over pre-existing tidal flat or tidal marsh plain (marsh peat or bay mud underlie pond substrates) and extending over drowned (hypersaline flooded) portions of terrestrial alluvial fans.

**Hydrology.** Tidally flooded by over-marsh flows during spring tides, storm-elevated tides, wave overtopping of marsh-fringing barrier beach or berms. Evidence of dry season evaporative concentration of brine, evaporation-driven drawdown, emergence of hypersaline flats and brine pools in relict channels and depressions of pre-existing marsh. Subject to intermittent infrequent freshwater inputs from rainfall, overland flow, small stream, or groundwater discharge at landward edge. Moderately shallow and persistent flooding for months; No extant examples; historic only, but recommended for reconstruction of naturalistic facsimiles in 1999 Goals Report.

**Slopes.** Very low, near zero.



**Figure 4.2.8.** Natural salt pond. Excerpt from U.S. Coast and Geodetic Survey sheet T-635 (early to mid-1850's field mapping).

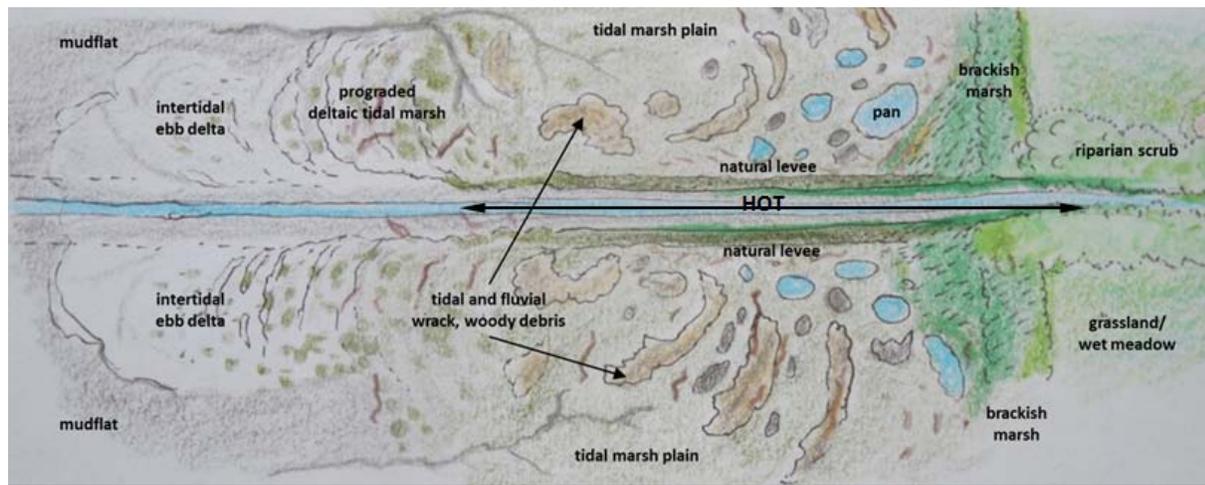
Vegetation. No vascular plants; hypersaline microflora: ubiquitous *Dunaliella salina*, cyanobacteria, halobacteria; high tidal marsh vegetation at pond margins; Terrestrial vegetation types at backshore: alkali grassland, vernal pool grassland, lowland grassland; local sedge-rush meadow.

Type Localities. None in present-day Estuary; partial modern analogs of derelict industrial salt concentrator ponds or managed salt ponds with summer drawdown and shallow brine.

## River or Stream



**Figure 4.2.9.** River or stream T-zone in profile



**Figure 4.2.10.** River or stream T-zone in plan view.

### *Narrative Description of Natural Physical Condition and Setting*

**Landscape position.** Associated with the discharge of a river, stream, or major outfall entering the Bay, extending from the bayward limit of the freshwater effects of the discharge to the upstream effects of the tides.

**General description.** Riverine T-zone length varies inversely with channel slope and varies directly with annual mean flow; width varies directly with frequency of out-of-channel flooding; can involve fluvial levees and associated riverine riparian vegetation that extend bayward as terrestrial peninsulas into tidal marshland.

**Substrate.** Channel bed consists of bay mud interspersed with fluvial deposits, which sometimes occur as annual strata. Terrigenous bedload ranges from clays and silts to gravels, although many urban channels are armored with boulder (rip-rap) or concrete rubble.

**Slopes.** Bed within T-zone mostly not steep (<1%) steepness often increasing near upstream limit of the Head-of-Tide (HOT); stream banks generally steep (>30) due to channel incision or because banks are revetments, riprap, or artificial levee faces (natural fluvial levees much less steep).

**Hydrology.** Mixed tidal and fluvial, with fluvial regime being perennial, ephemeral, or episodic. Perennial regimes are strongly seasonal unless influenced by POTW discharge.

Terrestrial vegetation types at tidal marsh edge.

- high salt and brackish marsh
- riparian can include native and ornamental trees and shrubs.

Type Localities. (1) San Antonio Creek (Sonoma/Marin Counties); (2) Sonoma Creek; (3) Coyote Creek



**Figure 4.2.11.** Type localities of riverine T-zone. Inset photo shows floating debris accumulating at Head-of-Tide.

# Science Foundation Chapter 5

## Risks from Future Change for Wildlife

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## INTRODUCTION

### Chapter Orientation

Recommendations for effective management actions require in-depth and integrated understanding of climate change impacts on wildlife. In this chapter, we include species of both plants and animals as ‘wildlife.’ Here we address two over-arching questions:

- How will populations of plants and animals in bayland habitats be affected by climate change?
- Which management actions can be most effective in keeping populations healthy and/or restoring population health, given anticipated climate change impacts?

Climate change is not a new form of stress but an intensification of impacts wildlife have suffered from human use since the Gold Rush: habitat loss and fragmentation, impaired ecological processes associated with land and water use, invasive and pest species, reduced and isolated populations, reduced population fecundity, reduced dispersal opportunities, etc. For both human populations and wildlife, many of the

greatest impacts of climate change will occur suddenly rather than gradually – storm surges that overtop levees and inundate human infrastructure will simultaneously have disastrous effects on wildlife. Wildlife protection must address both the long term trends in stresses and the sudden catastrophic events. We emphasize the dynamic nature of the environment for wildlife – past, present, and future – as well as the functioning of wildlife within the changing environment. All wildlife in the San Francisco Estuary have evolved to accommodate environmental change. In the past millennium there have been strong fluctuations in salinity and in rates of sea-level rise, (see Science Foundation Chapter 2). The difference now is the multitude of stressors on wildlife and the impaired ability of wildlife populations to respond to these stresses. For instance, housing developments in areas adjacent to intertidal habitat eliminate refuge habitats during times of high water and thereby promote excessive predation on marsh wildlife. Therefore, in this chapter we consider how resilience of wildlife to climate change and other stressors can be maintained and enhanced. With climate change and other changes in land-use, the landscape – that is the mosaic of habitats in the San Francisco Estuary – will change and organisms must respond to that change. In many cases, a species can thrive only if it is able to move, to colonize new habitat, or re-colonize areas formerly occupied. Therefore we emphasize management actions that promote successful dispersal and recruitment into new areas, in addition to promoting persistence in areas currently inhabited. In this way, actions that maximize resilience are most likely to be effective.

A healthy baylands ecosystem is characterized by heterogeneity rather than uniformity. A mosaic of habitat patches allows an array of species to persist, but only if the mosaic components are functionally connected. Plants and animals must be able to successfully move from one patch to another, at short (e.g., daily) or long (annual, decadal) time scales. The goal is a landscape that is dynamic and responsive; flexible, rather than static. The desired landscape is one that changes with time in a way that maximizes the long-term persistence of wildlife populations in the face of climate change. The original Bayland Goals Report (1999) was primarily concerned with targeting maps to achieve a mix of habitats in an estuary in which many important habitat types were missing or under-represented. This 2015 update for climate change emphasizes persistence of plants and animals in areas currently inhabited, as well as successful dispersal and recruitment into new areas, many of which did not exist in 1999.

## Approach

We emphasize **population-level** consequences of climate change on wildlife. That is, climate change effects on long-term **population trends** and **population viability** or **sustainability** of bayland plants and animals. These population trends depend on survival, reproductive success, recruitment and dispersal rates. To understand and address the impacts of climate change on population viability, we focus on **population resilience** – by which we mean how well various species tolerate or recover from changes in the environment. We focus on population-level consequences while also stressing the importance of habitat. Bayland habitats are critical in terms of the plants and animals they support, but habitat change is only one of the impacts of climate change on wildlife. Adequate habitat is a necessary but not sufficient requirement for long-term maintenance of wildlife in the face of climate change.

Survival, reproductive success, recruitment, and dispersal rates are difficult to measure, and even more difficult to predict. We qualitatively describe how climate change will affect these key population processes, based on the insights and predictions we have available. We use a conceptual model and draw on ecological models to describe these processes.

We are concerned with both **migratory animals** and **year-round residents**. For migratory species, survival during their time in baylands may, in some cases, be affected by climate change just as it is for year-round residents. In other cases, climate change effects for migrants in the estuary can alter survival and/or reproductive success elsewhere, i.e., on the breeding, feeding, or wintering grounds. In contrast, for other populations of migratory animals, conditions elsewhere may override any effects of conditions in the estuary.

Our approach integrates diverse **environmental processes** on **population trends and viability**. We use case studies of individual species to describe the effects of drivers on population trends and viability. We briefly describe how those drivers interact. We then look across the case studies for patterns of impacts across species. With that background, we provide a basis for management actions that will enhance population resilience and thereby maintain or restore wildlife populations.

The chapter does not describe all possible ecological effects of climate change on plants and animals, but rather those that can be expected to most affect population maintenance, resilience, etc., and thus require management attention. A change in **phenology** (i.e., the change in timing of important events in the annual cycle), for example, is not by itself deleterious, unless such a change results in increased mortality, decreased reproductive success, etc. In some cases, climate change may lead to a mismatch in timing between a predator and its prey; that may lead to, say, a decrease in body condition, which then may translate into increased mortality or decreased reproductive success. We evaluate the evidence for such impacts and linkages. Similarly, climate change may result in a change in **species co-occurrence** as species alter their habitat use (Ackerly et al. 2012); this may in turn increase competition, predation or disease and, thus, alter survival, reproductive success, recruitment, or dispersal.

## Framework

Impacts of climate change will depend on: a) the magnitude of the change in stressor, b) the sensitivity of a species, and c) the resilience of the organism, i.e., its ability to overcome or compensate for change due to stressors. A climate change impact may be of concern because the change is of great magnitude or because of high sensitivity to a change in the stressor. The impact will also depend on resilience of the population, which can depend on the effects of other stressors or the degree to which the population has recovered from previous stressor impacts.

### 1) **Uncertainty and Risk**

The consequences of climate change for plant and animal species cannot be predicted with certainty. Climate projections themselves show much variability and uncertainty, and new projections are provided regularly, superseding previous projections. We rely on summaries from other workgroups (see Science Foundation Chapters 1, 2, and 3) as well as other recent reviews (e.g., Cayan et al. 2012, Heberger et al. 2012, Ackerly et al. 2012) for these projections. The **consequences**, i.e., the effects of such changes (both positive and negative) for wildlife, add another layer of unpredictability. For example, changes in mean precipitation are difficult to predict for the San Francisco Estuary and its watershed, but what is clear is that precipitation patterns will become more variable (on daily, seasonal, and annual scales); that is, extreme precipitation events will become more common (Cayan et al. 2012), leading to increased probability of droughts and floods. Hence, a probabilistic framework is needed and management approaches must be designed with this unpredictability in mind.

**Risk** provides a useful concept for this chapter. The National Climate Assessment (Burkett and Davidson 2013) partitions risk into the product of **consequence** and **likelihood**. Consequence reflects **vulnerability to climate change** and other stressors. Likelihood reflects **probability of occurrence**. A severe outcome of low probability can be as much a concern as a moderate outcome with intermediate probability.

This approach, by itself, is well-suited for consideration of infrastructure. However, living systems are much more dynamic. Plants and animals have adapted to environmental change, rather than environmental stasis. The San Francisco Estuary has been especially dynamic, and has changed radically in the past 10,000 years (Parker et al. 2012a). Local extinction and dispersal to, or colonization of, new habitat has occurred repeatedly in the past. What is different now are the cumulative, anthropogenic effects such as (1) habitat fragmentation and alteration, barriers to dispersal (such as freeways, shopping malls), (2) contaminants, and (3) alteration of habitat and foodwebs due to non-native species introductions. Moreover, the rate of climate change anticipated in coming decades is unprecedented. The combination of increased rate of change, greater extreme events, and additional stressors, poses a high risk to wildlife. We emphasize that living species have some capacity to tolerate or adapt to climate change. It is the combination of increased climatic impacts with ongoing human impacts that threaten wildlife populations. Management actions need to **enhance resilience of plant and animal populations**. In particular, this means reducing mortality on adults and juveniles, increasing reproductive success, ensuring their ability to disperse, while maintaining genetic diversity. Thus, the ability to withstand impacts of climate change depends critically on managing the effects of all stressors. Management actions may be more effective with regard to co-acting stressors, compared to actions that address climate change effects directly. We emphasize the importance of episodic **extreme events**. The challenge is to consider relatively rare, catastrophic events that have profound long-term consequences (Callaway and Zedler 2004). One extreme event can lead to local extinction, which provides a reason to consider **re-colonization** potential. A site may lose its wildlife population but be suitable to re-colonization. To that end, we require “source” populations in proximity, and facilitate successful dispersal. Current conditions (such as habitat fragmentation, alteration, and contaminants) have already exerted deleterious effects and thus increased vulnerability to climate change.

## 2) Case Studies

The basis of this chapter is a suite of 32 case studies covering a wide variety of plants and animals. Our intent is not to duplicate the extensive case studies published in 2000 for the Goals Report (Goals Project 2000), but rather to update them in three ways, by considering: (1) the likely impacts of climate change, (2) the changes observed for species and information learned since 2000, and (3) specific management recommendations relevant to (1) and (2). We include two first-time case studies, on tidewater goby and grunion, because changed conditions in the estuary, including climate change, may allow these extirpated species to return. Some case studies combine similar species into suites of species that are likely to respond similarly, or for which we do not have enough information to justify separate entries. We refer to select case studies written for the Bay Interface workgroup (Science Foundation Chapter 3), as well as by our own workgroup members; all case studies are available in Appendix 5.1.

The workgroup used three primary criteria and two secondary criteria to choose species.

Primary criteria were:

- Ecological processes and status of the species are well understood
- High conservation concern and/or especially vulnerable to climate change
- Representative of other species.

Secondary criteria were:

- Species or species group especially associated with bayland habitat.
- The species or species group plays an important ecological role (e.g., with regard to foodweb).

Our intent is address threatened and endangered species as we encompass the broader community of bayland plants and animals of concern. The 32 case studies are listed in Table 5.1, with information on broad taxonomic group, sub-category within that group, and habitat.

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## CONCEPTUAL MODEL

Our conceptual model (Figure 5.1) complements the habitat-focused models presented in Science Foundation Chapters 1, 2 and 4.

For wildlife, we are concerned with two main types of climate change impacts, long-term trends and episodic events. At the top of the figure are three classes of drivers:

1. Trends in ocean conditions, ocean chemistry, tidal range, and sea level. These will produce changes in inundation regime, salinity, and chemical processes in wetlands.
2. Trends in temperature, affecting ocean, runoff patterns, and local conditions. These will change sediment supply, runoff patterns and chemical and biological processes in wetlands.
3. Changes in the intensity, frequency, duration, and/or timing of unpredictable but severe events, especially floods, droughts, and thermal stress, which in turn affect wildlife. These unpredictable events can produce immediate, direct and long-lasting impacts on wildlife populations that increase the risk of extirpation and extinction.

Habitat change is mostly driven by the trends in the first two classes of drivers. Even levee breaks or scarp collapses that suddenly change habitat features are primarily the result of ongoing trends of sea level rise, sediment starvation, etc. On average, plant and animal populations will reflect long-term trends and associated changes in habitat structure and average salinity distributions. But average population size through time is not the best indicator of population viability or sustainability. Extinction risk particularly reflects extreme events, floods, droughts, and storms (Thibault and Brown 2008). Such risk is amplified by smaller and scattered populations that may result from habitat fragmentation and loss. Habitat loss and fragmentation also strongly affect an organism's ability to rebound from the effects of extreme events.

Thus, our conceptual model addresses both gradual and stochastic (i.e., unpredictable) impacts of climate change on wildlife populations, as well as the interactions among the various stressors. These stressors already affect plants and animals—here we elucidate how climate change interacts with additional stressors.

### Trends and Episodic Events

Changes in habitat, and its spatial context, are displayed on the left side of Figure 5.1. **Depth** and **elevation** of wetland habitats and the **salinity** and **turbidity** of the overlying aquatic habitats will change in response to long term changes in sea level, sediment flux, etc. Interannual variation in weather will cause such features to change more in some years than others, but overall climate change will affect ongoing

trends. These trends in physical parameters will produce corresponding trends in the **size** and **connectivity** of habitats for most species. Habitat quantity, quality, and connectivity are all fundamental drivers with respect to the long-term population trends, abundance, and resilience of every plant and animal species.

Climate change will affect habitat in two ways: (1) configuration of habitat in the larger landscape matrix as reflected in changes in **connectivity** and **size**; (2) suitability of habitat needed to support healthy, resilient populations (e.g., contain suitable nest locations, foraging opportunities, refugia from predators, etc.), which includes **patch configuration**. Changes in habitat will affect not only the ability of populations to persist in an area but in their ability to **colonize** newly suitable habitats or **recolonize** suitable habitats when their populations recover from disasters. Thus, changes in patch configuration will have large impacts on species with limited range or large range requirements, limited dispersal abilities, and species which are already suffering from habitat loss and fragmentation. In this chapter we use the case studies to identify the types of sensitivities for each species.

The effects of episodic, extreme events such as severe storms, floods and droughts are illustrated along the right side of Figure 5.1. The **degree, timing, duration** and **frequency** of such events can all reduce the **survival** and **fecundity** of plant and animal populations. Each parameter of extreme events can have different effects on the wildlife populations of the baylands; a brief and rare extreme event can have different impacts than a less extreme event that is frequent and prolonged. The impacts of events can interact; droughts in California have often been ended with a flood year – so a population that is reduced by drought must then cope with the impacts of flood. Increased frequency of extreme weather event will reduce the time available for a population to recover before the next extreme event occurs. Such events are expected to be both more frequent and more intense (Flick et al. 2003, Dettinger 2011, Cayan et al. 2012).

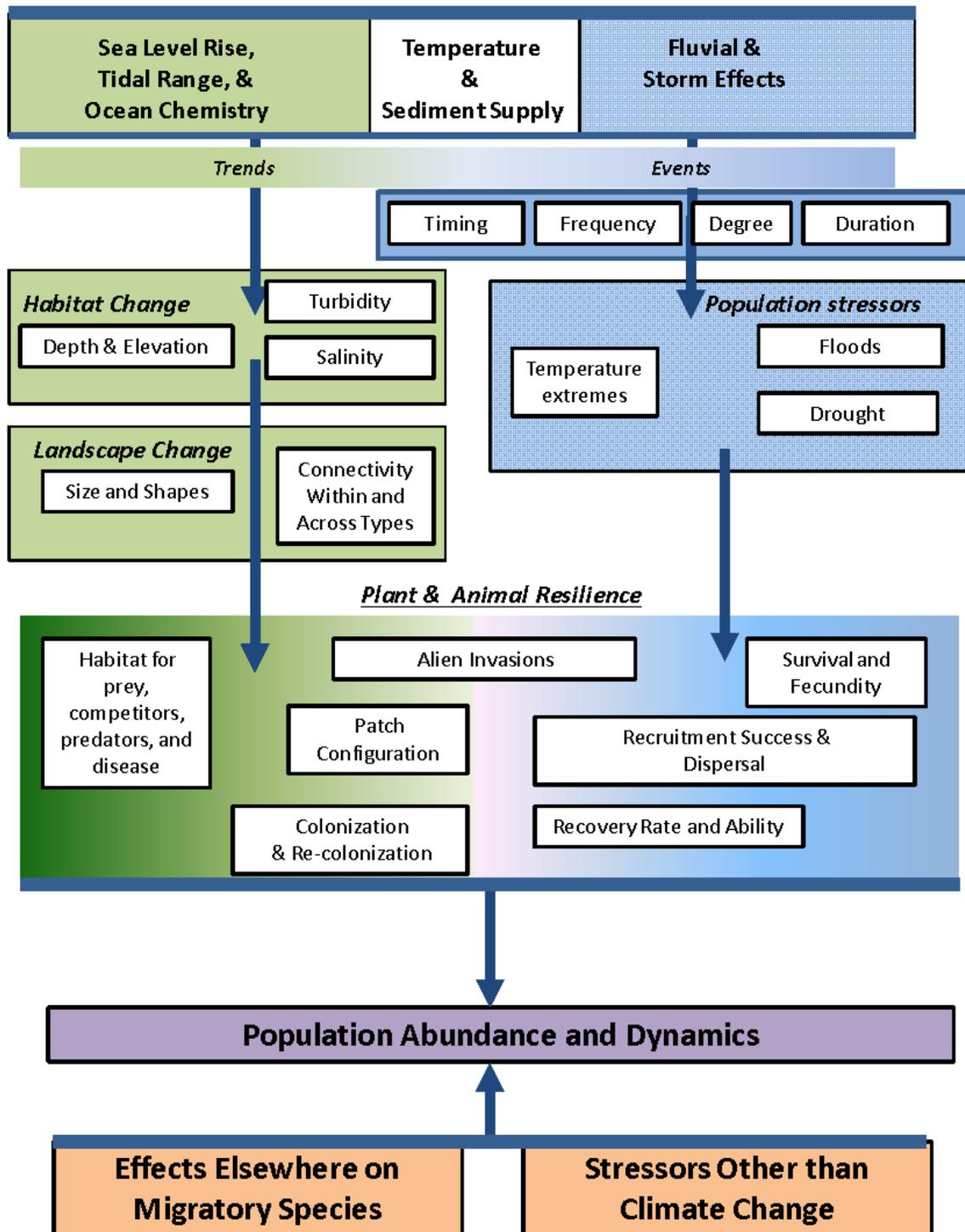
Episodic events test the resilience of wildlife populations. A single extreme event can have serious consequences for a local population (Zedler et al. 1989, Callaway and Zedler 2004). Extreme events may directly reduce the **survival** of members of the population; extreme events may reduce nesting success or feeding success thus reducing **fecundity**. Reductions in survival and fecundity will reduce the **recovery rate and ability** of the population. If the degree or frequency of catastrophic events exceeds the capacity of a population to recover then the population is at risk of extinction. If populations cannot reestablish themselves (following local extirpation) the species is at risk of extinction. Species at greatest risk will be species with low fecundity, limited dispersal effectiveness, and/or high mortality due to anthropogenic or other impacts. Species of already reduced population size because of ongoing or previous stressors are at enhanced risk.

The exact nature of episodic stressors must be defined with respect to the sensitivities of individual species. The temperature tolerances of one wildlife species will determine whether or not a particularly hot summer will decimate the population. Conversely, extensive and diverse habitats are likely to provide more thermal refugia for a given species and thereby reduce the impacts of extreme events. Long-term trends can also have important impacts on the long-term average abundance of a population that render them less resilient overall. Our conceptual model emphasizes that species are affected by both year-to-year variability in conditions as well as by long-term trends.

**Table 5.1.** Summary of case studies.

General Category	Species	Sub-category, i.e., what is it indicative of	Habitat	resident vs. migrant
<b>Mammals</b>	salt marsh harvest mouse	marsh (tidal and non-tidal) small mammal	tidal marsh; diked bayland	resident in baylands
	Suisun shrew, salt marsh wandering shrew	marsh (tidal and non-tidal) small mammal	tidal marsh; diked bayland	resident in baylands
	river otter	aquatic mammal (creeks and rivers)	creeks and rivers	mostly terrestrial/bayland interface
	harbor seal	aquatic mammal, using bay and mudflat	open bay, mudflat, sandbar, rocky inter-tidal	resident in baylands
<b>Marsh Birds</b>	Ridgway's rail	tidal-marsh dependent birds	tidal marsh	resident in baylands
	song sparrow	tidal-marsh dependent birds	tidal marsh	resident in baylands
	black rail	tidal-marsh dependent birds	tidal marsh	resident in baylands
	northern harrier	marsh predator	multi-habitat	resident, multi-habitat
<b>Water Birds</b>	American avocet, western sandpiper	avocet: large shorebirds; sandpiper: small shorebird;	marsh; mudflats; managed pond	avocet; breeder in baylands; sandpiper: migrant;
	least tern and Forster's tern	fish-eating birds	beaches, marshes, sloughs, islands	breeder in baylands
	dabbling ducks: northern shoveler, northern pintail, American wigeon, gadwall, mallard, green-winged teal	six species of dabbling ducks included	diked bayland and tidal marsh; managed ponds	both resident and migratory species
	diving ducks: scaup (lesser and greater), surf scoter, bufflehead, canvasback, ruddy duck	bay ducks; sea ducks; stiff-tailed ducks	diked bayland; open water; managed ponds	predominantly migrant
<b>Herps</b>	California toad	wetland amphibian	wetlands	resident
	California red-legged frog	wetland amphibian	wetlands	resident
<b>Fish</b>	Pacific herring	subtidal	shallow aquatic	migrant
	delta smelt	upstream part of estuary	open water	migrant
	longfin smelt	pelagic throughout Estuary	open water	migrant

	longjaw mudsucker	marsh fish	pickleweed marsh	migrant
	tidewater goby	small estuaries	estuarine lagoon	breeder
	grunion	recovered native	sandy beach	breeder
	chinook salmon and steelhead	migratory fish	vegetated marsh edge	migrant
<b>Invertebrates</b>	Macroinvertebrate: Dungeness crab	Aquatic: nursery value of baylands	shallow aquatic, eelgrass	migrant
	terrestrial marsh invertebrates	multiple species	tidal marsh	resident
<b>Vernal Pool</b>	plants, crustaceans, other invertebrates, plants	multiple taxa	freshwater, ephemeral pools	resident
<b>Plants: <i>Spartina</i></b>	invasive <i>Spartina</i>	invasive and native <i>Spartina</i>	tidal marsh	resident
<b>Plants: rare and important plants</b>	submersed aquatic vegetation	multiple species	open water	resident
	low tidal marsh graminoids	multiple species	tidal marsh	resident
	high tidal marsh annual forbs & graminoids	multiple species	tidal marsh	resident
	high tidal marsh subshrubs and perennial forbs	spring high tide zone	tidal marsh	resident
	high tidal marsh perennial graminoids	spring high tide zone	tidal marsh	resident
	terrestrial ecotone/high marsh graminoids	multiple species	terrestrial ecotone (transition zone)	resident
	terrestrial ecotone psammophyte	multiple species	terrestrial ecotone (transition zone)	resident



**Figure 5.1.** Conceptual model of climate change impacts on bayland wildlife. The left (green) side traces the impacts of long-term trends on the habitats and landscape occupied by wildlife. The right (blue) side traces the impact of episodic, extreme events and other factors that will affect population processes, and thus population size, trends, and resilience. Temperature affects both sides. Combined, these factors determine the abundance and dynamics of wildlife populations.

## Combined Patterns

In the center of Figure 5.1 are three aspects of climate change impacts on wildlife that operate both as episodic, unpredictable events and long term trends: **temperature, sediment supply, and alien invasions.**

Climate change is expected to raise temperatures in California and thereby produce less snow in the mountains and more rain. Thus, even with no change in total precipitation, the timing, duration and degree of outflow will change on the short time scale, while on the longer time scale we expect increased frequency, degree and duration of droughts and floods (Science Foundation Chapter 1. Changes in air temperature in the estuary will have local effects on the reproduction and survival of many plants and animals; in many cases, changes in plant growth and reproduction will have secondary impacts on the animals that depend on them.

Sediment processes have significantly changed in the estuary as described in Science Foundation Chapter 1. Gold mining, water management, urban development and dredging have all affected the sources and supplies of sediments in the estuary. Future water development, habitat restoration, climate change, and catastrophic levee failure all have the potential to shift sediment dynamics in the estuary both gradually and suddenly. Impacts of such change are difficult to predict and are likely to vary in different parts of the estuary. We have relied on the projected habitat changes described elsewhere (Science Foundation Chapter 2 and its Appendix 2.1) and only here briefly summarize expected habitat change reflecting sedimentation. Invasion by alien species is an excellent example of an episodic event, often traceable to a single date or event for each successful invasion. However, long-term changes in habitat are often an important precursor to successful invasion. The responses of humans and wildlife to climate change may facilitate invasion by new species in the estuary. Such invasions will include new diseases, competitors, predators and prey, but may also release species that are currently present at low numbers to become new dominant species (as happened with *Egeria* and largemouth bass in the delta over the last 20 years; Santos et al. 2011).

## Climate Change Impacts in Context

For all species the impacts of climate change constitute only one set of stressors they must deal with. Shifts in **stressors other than climate change** may exacerbate the impacts of climate change or affect the resilience of populations to respond to such impacts; both need to be considered.

For migratory species, conditions elsewhere (in the ocean, in the arctic, in Mexico, etc.) may override any efforts we make to protect them while they are present in the estuary. Conversely, changes in the ecology of migratory species elsewhere may make the baylands here more important than before.

The biology of bayland wildlife populations have been strongly affected by harvest, habitat change and California's exceptionally variable weather. Many of our aquatic and avian wildlife species are strongly affected by conditions outside the bay, either in the ocean or on summering and wintering grounds. Climate change will alter the impacts of both human impacts and weather and produce significant changes in the **population abundance and dynamics** of many valued species.

## FUTURE CONDITIONS AFFECTING WILDLIFE

As part of the Bayland Ecosystems Habitat Goals Update project, five principal climate change scenarios have been delineated (Chapter I Science Summary). Here we consider the short-term and long-term impacts of such changes for wildlife.

### Future Scenarios of Climate Change and Impacts for Wildlife

We summarize the five climate change scenarios considered in the Bayland Goals Update Project, which we refer to as Scenarios 1-5. Note that the first four scenarios considered combinations of either low sea-level rise (0.52 m over 100 years; Scenarios 1, 3) or high sea-level rise (1.65 m over 100 years; Scenarios 2, 4) and either low sediment concentration (Scenarios 1, 2) or high sediment concentration (Scenarios 3, 4). Each of these 4 scenarios in turn allowed for two sets of projections with regard to temperature, precipitation, snowmelt, runoff, and salinity: the **Ga** projection (much warmer and drier) and the **Pb** projection (moderately warmer, but no change in precipitation). Both sets of projections posit that future conditions will include:

- Warmer air
- Higher salinity
- Decline in snowmelt contribution to runoff, with a possible decline in precipitation and runoff
- Earlier runoff in winter and thus less water in late spring, summer
- Possibly lower suspended sediment concentrations (not likely to increase)

Of special concern for wildlife, the scenarios project:

- Increasing frequency of extreme environmental conditions such as higher water temperatures, higher storm surges, higher flood peaks, and possibly droughts

Predictions regarding future habitat extent and configuration differ strongly among Scenarios 1-4. To facilitate comparison, here we focus on the two most extreme of the first four scenarios: Scenario 2 (High sea-level rise, low sediment concentration) and Scenario 3 (Low sea-level rise, high sediment concentration); results for Scenarios 1 and 4 are intermediate with respect to 2 and 3.

The habitat projections of the four Scenarios are of particular concern with regard to tidal marsh habitat, which is divided into low marsh, mid-marsh, and high marsh, as well as tidal (or mud) flats. Different species depend on the three different tidal marsh sub-habitats (as described in the case studies and in this chapter). However, the total amount of tidal marsh habitat is also a fundamental concern, and so we summarize “combined marsh” as well, which includes all three marsh sub-habitats. We do so for three time periods: from 2010 to 2030; to 2050; and to 2110.

An important restriction on these projections is that they do not include **specific** restoration efforts which have been implemented since 2010 or will be in the future. Instead there are two projection modes: either (1) no restoration is assumed or (2) restoration in diked baylands occurs everywhere—that is, all levees are assumed to have been removed as of 2010, allowing restoration to occur. However, the actual time course of restoration is not modeled. Therefore in the following summaries we consider model output with regard to two extremes: no restoration and complete restoration everywhere. For change to 2030 and to 2050, we consider only “no restoration”. That is, model output depicts what would be expected due to changes in

elevation and salinity alone, without assuming specific restoration outcomes. For 2110, we present comparable results as well, but in addition we also summarize change in habitat if all levees were removed as of 2010 and assume that by 2110 restoration was complete.

**2010 to 2030 (No restoration):**

For Scenario 2 (**Hi SLR/low sediment**):

- increase in mid marsh (+26%) and low marsh (+14%); slight increase in mudflat (+2%). loss of high marsh (-54%).
- note that combined marsh increases 8%.

For Scenario 3 (**Lo SLR/high sediment**):

- increase in mid marsh (+73%). loss of low marsh (-32%), high marsh (-32%) and mudflat (-46%).
- note that combined marsh increases +29%.

Thus, mid-marsh habitat increases and high-marsh habitat decreases under both scenarios, but projections regarding mudflats differ: either little change or substantial decline.

**to 2050 (No restoration):**

For Scenario 2 (**Hi SLR/low sediment**):

- increase in low marsh (+95% compared to 2010), decrease in mid-marsh, decrease in high marsh. For mid-marsh, though, the net loss is only -4%. By 2050 -75% net loss of high marsh (without considering any additional restoration). Mudflats continue to increase (+16% comparing 2050 to 2010).
- Combined marsh is +5.8% net increase for 2010 to 2050.

For Scenario 3 (**Lo SLR/high sediment**):

- increase in low marsh (+4% net increase from 2010 to 2050, following decrease from 2010 to 2030), increase in mid-marsh (net increase of +101%), continued decrease of high marsh (net decrease of -54%). Mudflats also continue to decrease, strongly (-80% comparing 2050 to 2010).
- Combined marsh shows continued increase (+49% net increase).  
Thus, total marsh habitat increases (from 2010 to 2050) but high marsh decreases substantially under both scenarios. Projections regarding mid marsh and mudflat differ, depending on Scenario.

**to 2110 (No restoration):**

For Scenario 2 (**Hi SLR/low sediment**):

- Compared to 2010, very significant decrease in mid marsh and high marsh (-94% decrease or greater). Increase in low marsh (38% if we look only at current tidal, no additional restoration). Mudflat continues to increase (net +158%).
- There is 63% decrease for combined marsh compared to current tidal marsh.  
In short, low marsh increases, mudflats increase, others decline.

For Scenario 3 (**Lo SLR/high sediment**):

- Low marsh shows strong decrease (from +4% increase by 2050 to net -65% decrease in 2110). Mid marsh continues to increase (net increase from 2010 to 2110: +158%). High marsh continues to decrease (net change, -80% comparing 2110 to 2010). Mudflats continue to decrease (net -91%).
- Combined marsh continues to increase to +61%.  
In short mid-marsh is a big winner; low and high marsh decrease, as do mudflats.

**to 2110 (complete restoration):**

For Scenario 2 (**Hi SLR/low sediment**):

If all levees are removed, resulting in full restoration by 2110, there would be a slight increase in combined marsh compared to the present tidal marsh (net +1.7%). Nevertheless, there would still be strong decrease in mid marsh and high marsh (-42%, -57% respectively). Mudflats are expected to increase (net +33%).

For Scenario 3 (**Lo SLR/high sediment**):

If all levees are removed resulting in full restoration by 2110, there would be a large increase in combined marsh compared to the present tidal marsh (net +327%). There would be strong increases in low and mid marsh (+129%, +523% respectively) compared to the present. For high marsh, though, there would still be -26% decrease compared to present tidal high marsh. Mudflats are expected to decrease (net -64%).

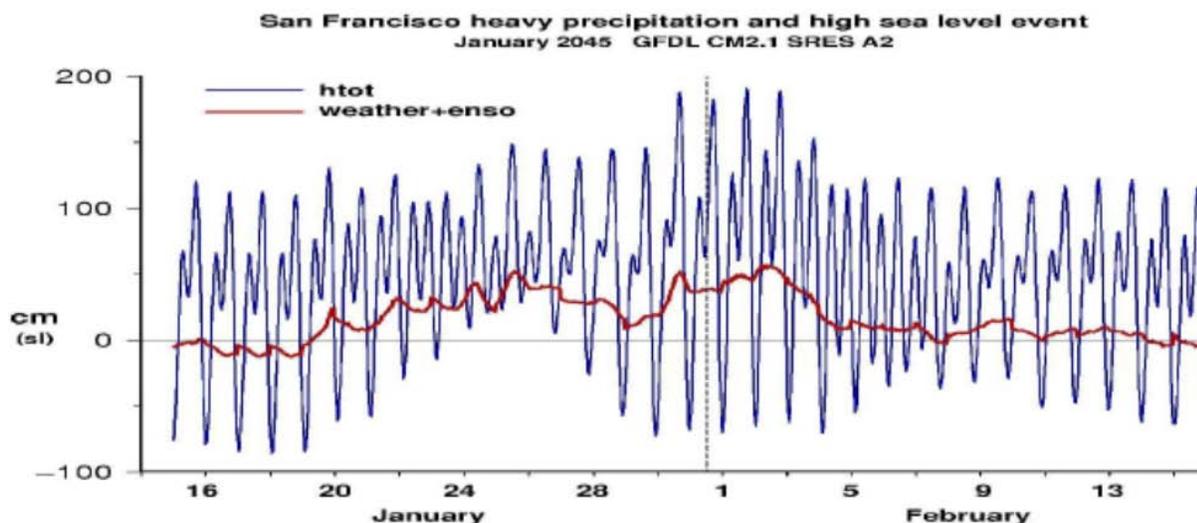
In short, model projections indicate loss of high marsh under all Scenarios; the change in mid marsh and total marsh depends on the Scenario, but note that under Scenario 2, the net habitat change is more than 90% loss for mid and high marsh by 2110.

The above synopsis focuses on the total amount of habitat in the estuary. We must also consider sub-regional changes (see Science Foundation Chapter 2, Appendix 2.1) as well. Another important implication of these models is that habitat will change in a fundamental fashion from what it is now. Current wetland habitat, whether low, mid, or high marsh, or mudflat, will not as a rule remain as it is, at least not in the long-term. **Thus, the ability of organisms to successfully disperse to and colonize new habitat is essential.**

Scenario 5 concerns an extreme event: a hypothetical winter storm associated with flooding and extended inundation. The particular example modeled is that of a late January storm lasting for 2 weeks, but whose effects last even longer. Astronomical high tides at this time of year, El Niño, and the storm all contribute to extreme high water levels (Figure 5.2). Wind waves and changes in salinity can be expected. One of the most significant aspects of such a storm is extensive inundation of habitat, which may affect survival of tidal marsh wildlife (if there are no adequate refugia), and would likely reduce ability to forage for many animals (Thorne et al. 2013). While this scenario is hypothetical, Thorne et al. (2013) document two such severe and long-lasting events in San Francisco Bay tidal marshes: one in January 2010 and a second in March 2011. In the latter event, inundation nearly tripled compared to the non-storm period; such long periods of inundation will strongly affect many plant and wildlife species, as we discuss in this chapter.

### The future of non-tidal, managed habitat

As important as intertidal habitat is, we must also be concerned with diked bayland habitat, which currently composes a substantial proportion of bayland habitat. These are low-lying areas around the Bay that were once tidal but now receive very little to no tidal action. There are several types of diked bayland habitat and some are especially valuable to wildlife. Of particular note are **managed ponds**, a category which includes both “salt ponds” (ponds that currently produce or in the past produced salt) and storage/treatment ponds. In this chapter we are particularly concerned with those managed ponds that, to a greater or lesser extent, are operated to maximize the value to wildlife. These are constructed habitats within the Bay, but at present managed ponds play a vital role for many wildlife species, such as shorebirds and waterfowl. Salinities of these non-tidal habitats are changing and will continue to change, especially as current managed ponds are converted to restored tidal marsh. Reductions in salinity of managed ponds will strongly impact prey availability for many waterbird species (Takekawa et al. 2009).



**Figure 5.2.** Scenario 5: Winter storm event. Scenario provided by D. Cayan (Scripps Institute). The Y-axis depicts fluctuation in water levels during the simulation (total water level in blue) with the portion due to the storm in conjunction with the ENSO event depicted in red.

### Physical Effects on Wildlife and their Habitat

Science Foundation Chapters 1 and 2 describe climate change impacts on the physical characteristics of the bayland ecosystems; Science Foundation Chapter 3 describes physical and biological effects climate change will have from the ocean side. Here we attempt to identify some of the ways that climate change will affect bayland plants and animals. We consider broad patterns common to many species as well as more specific issues raised in the various case studies.

We consider four major categories of environmental drivers (see Figure 5.1, Conceptual Model):

1. Ocean Effects: Long term trends in salinity, nutrients and tidal heights
2. Weather effects: Short-term events including storm surges, freshwater flows, contaminants, and suspended sediments.

3. Geometry: Changes in bathymetry and the nature of the land:water interface.
4. Temperature: Changes in aquatic and aerial thermal regimes.

### *Ocean Effects*

#### Salinity

Increasing salinity due to sea-level rise will directly affect most plants and aquatic animals, almost all of which have salinity as a principal determinant of their distributions. Fish communities of the estuary differ in species composition primarily due to different salinities in different areas. Strictly freshwater fish that currently can be found in Suisun Marsh will become rare. On the other hand strictly marine fish, including halibut, flounders and white seabass are likely to become more common components of the fish community of Central San Francisco Bay (Moyle et al. 2012). Amphibians require, at least at some life stage, freshwater conditions so they are likely to move upstream from some of their present locations. Plant distributions, persistence, and species composition will all be affected by changes in salinity and by the combined impact of salinity and inundation (see below; Parker et al. 2012b; Grewell et al. 2014). For plants, soil salinity is of particular importance, not simply the salinity of adjacent channels and the bay (Parker et al. 2012a).

Shorebirds and other waterbird species are strongly affected by changes in salinity, often acting through changes in their prey (Warnock et al. 2002, Takekawa et al. 2006b). Terrestrial birds and mammals are less directly affected by changes in salinity but changes in plant distribution are expected to strongly alter distributions of terrestrial vertebrates. Veloz et al. (2013) found that the projected change in salinity was the most important predictor of future population trends for tidal marsh birds in the San Francisco Estuary, presumably acting through projected changes in vegetation.

#### Water depth and inundation

Inundation of bayland habitat is projected to increase, due to both sea-level rise and to storm events (Cayan et al. 2012, Thorne et al. 2013, Nur et al. 2012). Long-term effects of sea-level rise on inundation patterns and water depths will be a function of geometry (see below) and substrate. Where inundation is not constrained by human infrastructure, the depth profile will move landward also. That is, erosion and sediment dispersion in newly inundated areas will reconstruct the typical gentle gradient of depths from marshes to sub-tidal areas. Where substrate or infrastructure will not allow erosion, habitats will be lost to excessive water depth as described in Science Foundation Chapter 1 (Knowles and Cayan 2002, Knowles 2010).

In areas with less adequate levees at present, particularly Suisun Marsh and the Delta, levee failure is almost inevitable and will result in inundation of many areas (Lund et al. 2007, Moyle 2008, Moyle et al. 2012, Moyle et al. 2014).

Dissipation of tidal energy into larger inundated areas near the mouths of the rivers draining the Central Valley will make the salinity gradient steeper and reduce the average area of suitable habitat for mesohaline species, like delta smelt. Such habitat alternation will be more pronounced in the

mesohaline waters of San Pablo and Suisun bays, than the more oceanic waters of the Central Bay or the lagoon-like waters of South Bay.

#### Other effects

Increased intrusion of ocean waters will bring other features of ocean water into the bay, including cooler, more nutrient-rich waters under most climate-change scenarios, and increased acidification. These factors are likely to be much less important to wildlife than changes in salinity. There is little evidence of nutrient limitation in bayland ecosystems. Ocean acidification effects on bay water chemistry are unclear (see Science Foundation Chapter 3), but may limit the ability of larval invertebrates to accumulate enough calcium for shell growth.

Wildlife species that can move inland to follow the change in mean salinity distributions will be exposed to different physical conditions, potential isolation from other habitats, and new sorts of stressors. Species with less dispersal ability will be subjected to changes in the habitats they occupy. Later in this chapter we detail some of the likely negative effects on particular species.

#### *Weather Effects*

Decreasing snowfall in California is the most notable historic impact of climate change, and is likely to intensify (Roos 1989, Knowles and Cayan 2002). Increasing demand for a shrinking amount of available precipitation will likely yield more aggressive water management in the Central Valley. Together, these effects will produce shorter, sharper outflow events in the winter and longer seasons of low flow, beginning earlier in the spring and lasting longer through the fall (Stewart et al. 2005). Organisms of thermal sensitivity (like salmon), or which require a minimal duration of wetness (like the California red-legged frog) are at risk.

Storms are projected to be more frequent and likely more intense (Thorne et al. 2013). Pacific coast storms have increased in frequency and intensity in recent decades (Graham and Diaz 2001) and there is reason to be concerned about additional increases as a result of climate change. Peaks in streamflow and floods are predicted to occur earlier (Stevenson et al. 2005).

Storms are associated with high winds and stressful environmental conditions, which may lead to direct mortality or to increased mortality as a result of restrictions on foraging activity or reproductive success. Storms can have catastrophic effects on tidal wetland habitat (Zedler et al. 1989, Hilgartner and Brush 2006).

Intense storms following prolonged periods of dry conditions will intensify the loads of contaminants mobilized by the first flushing flows in each year, in particular mercury (Bergamaschi et al. 2001, Takekawa et al. 2006b). This will be true on a large scale due to flows from the vast urban and agricultural watershed of the Central Valley, but may be at least as important in each of the smaller watersheds draining into the bay. Management of stormwater runoff to protect wildlife, particularly aquatic wildlife, is likely to increase in importance.

Occasional tropical-weather based events (the 'atmospheric rivers' of Dettinger [2011]), are likely to become more common and produce rainfall events of large volume. Such extreme events, as in 1983 and 1997, have had significant impacts on diverse wildlife from plants to tidal marsh song sparrows (Zedler 2010, Point Blue Conservation Science, unpublished).

Sediment transport will be altered by changes in the pattern of water flow, as well as by changes in the sources of sediments to the estuary. Sediment loads have been decreasing for decades due to dam construction and the reduction in sediment loads from gold rush activities (Schoellhamer 2011). Particle sizes of sediments have gotten smaller as watershed development and dam operations preclude the movement of larger particles. Sediment loads are apt to decrease further as both intentional and catastrophic levee breaches produce new sediment traps. On the other hand, rising sea levels will expose new areas to erosion in bay area water sheds which may mobilize coarser substrates that are useful to valued wildlife like grunion and tidewater gobies.

### *Geometry*

Changes in bathymetry will occur through changes in depths and shifts in sediments, as described in the chapters (Science Foundation Chapters 2 and 3). In some areas sea-level rise will lead to gradually increasing inundation. However, in Suisun Marsh and the Delta, sea-level rise and/or higher flood peaks are likely to cause levee failure, producing large and rapid alterations of the geometry of the baylands. This transformation of habitats, from managed wetlands to subtidal, will have immediate impacts on the resident species while removing intertidal habitats from the estuary in the long term. Inundation of deeply subsided islands will produce lasting changes in mean salinity as described above, but also significantly change the bathymetry and hydrodynamics of the upper estuary.

Changes in geometry will produce changes in the area, placement and quality of diverse wildlife habitats. Increased depth will most directly reduce the habitats of shorebirds and dabbling ducks, as well as other waterbirds (Galbraith et al. 2002, Takekawa et al. 2009; see below for further discussion of depth of managed ponds). The indirect effects of changes in the inundated area in the bay are likely to affect almost all wildlife populations.

### *Temperature (Aerial and Water)*

Temperature of estuary waters and air are predicted to rise (Ackerly et al. 2012, Cayan et al. 2012). Coastal ocean temperatures are predicted to fall as a result of greater upwelling off the California coast (Snyder et al. 2003); more upwelling will also result in a stronger thermal gradient as one moves away from the coast. The severity and duration of extreme temperature events, such as heat spells, are expected to increase (Cayan et al. 2012).

Increasing air temperature in Northern California has already produced long-term declines in precipitation as snow in the Sierra Nevada mountains (Roos 1989). Thus, despite uncertainty regarding the quantity of precipitation in Northern California, there is little doubt that it will fall more as rain and will more quickly flow downstream. Increasing demand for an increasingly limited amount of available water for human use will lead to more upstream storage and diversion. Thus, freshwater storm inflows to the bay will often be earlier in the water year (Stevenson et al. 2005) and of shorter duration, but probably greater intensity (Dettinger 2011). Annual inflows to the bay are likely to become a smaller percentage of total precipitation, especially outside of the storm season.

Warmer temperatures on the land will tend to steepen the thermal gradients from the ocean in summer months, but, even in winter months, storms are expected to be more severe, more frequent and of longer duration (Thorne et al. 2013). Sea-level rise will amplify the effects of storm waves on water courses and bayland margins (Cayan et al. 2012).

## Summary of Combined Physical Impacts on Wildlife

Estuaries are defined as enclosed areas where salt water from the ocean mixes with freshwater from the land. Wildlife populations in estuaries array themselves along salinity gradients, either because of their own salinity tolerances or due to the salinity tolerances of the species they rely upon. San Francisco Estuary faces a future with increasing salt water intrusion as sea-level rises and decreasing freshwater inflows as more precipitation comes as rain rather than snow. Thus, the waters that inundate baylands will become more saline and species that rely on low to medium salinity conditions will be stressed.

Estuarine ecosystems are strongly affected by tidal action that moves nutrients and organisms twice each day across different habitats. Tides expose sedentary plants and animals to strongly changing conditions each day, to which they are well adapted. Changes in the geometry of the bay, such as would follow levee failure in Suisun Bay or the Delta, will greatly reduce tidal action. More ocean intrusion, less freshwater flow, and greater inundated areas with reduced tidal action will cause most of the northern estuary to become more like the lagoon nature of South San Francisco Bay (Cloern et al. 1985). Species adapted to varying salinities and temperatures and inundation patterns will have less of an advantage over specialist species. Species that rely on water currents to assist their movements among habitats will have reduced ability to move from habitat to habitat or to colonize suitable habitats elsewhere. More stable salinities and consistently warmer waters may facilitate the invasion of more species or in some cases re-establishment of species not recently seen.

The future wildlife of San Francisco Baylands will reflect these interacting physical changes, the biological changes we describe below, and the effectiveness of management actions to protect wildlife.

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## IMPACTS TO WILDLIFE

Here we summarize impacts of climate change on wildlife populations, based on the 32 case studies (listed in Table 5.1). The individual case study findings are summarized in Table S1. We first provide a quick overview by considering the 32 case studies grouped into five major categories: Marsh resident species; managed wetland (including managed ponds) and tidal flat, migratory and resident; transition zone (upland interface); aquatic species, not resident or breeding in the Baylands; aquatic species resident or breeding in the Baylands. We then provide a more detailed accounting of how climate change will affect bayland species by considering specific demographic processes.

### Summary of Habitat Type

#### *Marsh Residents*

Case studies include: low marsh plants, high marsh plants (saline and brackish; 3 case studies), invasive and native *Spartina*, marsh plain invertebrates, longjaw mudsucker, Ridgway's rail (formerly clapper rail), black rail, song sparrow, Suisun shrew and salt marsh wandering shrew, salt marsh harvest mouse, and northern harrier. Main climate-change concerns are: loss of habitat due to sea-level rise, inundation of habitat during winter extreme tides and storms and during the breeding season coupled with lack of refugia, elevated predation due to human-associated predators (e.g., crows and domestic/feral cats) as well as increased access to tidal marsh by predators. Future changes in salinity and inundation regime will affect plants important to wildlife as well as rare plants. Tidal marsh habitat is very fragmented and therefore connectivity of habitat is a major concern. Some species can take advantage of diked wetland habitat for refuge, foraging, etc., (e.g., shrew species and salt marsh harvest mouse) and this should be considered.

#### *Wildlife in Managed Wetlands, Including Ponds, and Tidal Flat (Migratory and Resident)*

Case studies include: dabbling ducks; diving ducks; American avocet and western sandpiper; least tern and Forster's tern. Whereas some species use tidal marsh habitat (e.g., dabbling ducks), managed wetlands (especially managed ponds) is generally a preferred habitat, and thus future habitat loss is a concern due to planned habitat restoration. Changes in salinity and water depth are also of great concern. Mud flats are projected to be lost due to sea-level rise. Reproductive success is sensitive to environmental conditions, including mercury contamination. Species can be affected by shifts in prey, which may occur with climate change, but which are hard to predict.

#### *Transition-zone (Upland Interface) Species*

Case studies include: Terrestrial ecotone plants (2 case studies); vernal pool species; California red-legged frog; California toad; and river otter. Availability of fresh or brackish water is key; saltwater intrusion is a major concern. Reproduction is very sensitive to environmental conditions (e.g., to changes temperature, salinity). Grazing by cattle is a concern for vernal pool plants and animals. Many species have limited dispersal. Habitat is very limited and is fragmented and often not connected. Upland refugia are important for these species as they are for marsh species and for many waterbirds, especially during storms and flood events.

#### *Aquatic Wildlife Not Resident or Breeding in the Baylands*

Case studies include: Dungeness crab, Pacific herring, delta smelt, longfin smelt, chinook salmon and steelhead. These species live predominantly in the water column and so move into Baylands when tides or floods inundate them. Many use baylands as nursery habitats for their young. Year to year variation in salinity distributions can control which species are in a particular geographic area in a given year. Thus, climate change impacts on salinity will have substantial effects on these species. Several of the species can be expected to be strongly affected by conditions away from the bay (i.e., oceanic condition). Some species are widespread and others of very limited distribution. Some species, particularly salmon and steelhead must migrate through the entire estuary on their way to the sea; in some cases their speed of movement will be determined by their physiological condition as they leave their native streams. Other species are subtidal residents that extend their range into Baylands, or can be expected to use Baylands as more of the Baylands become subtidal with sea-level rise.

#### *Aquatic Wildlife Resident or Breeding in the Baylands*

Case studies include: submerged aquatic vegetation, tidewater goby, grunion, and harbor seals. Aquatic species that rely on baylands for their breeding suffered badly through the middle of the last century before environmental protection began to reverse the degradation of aquatic habitats. Some of these were extirpated and have not yet returned but climate change and environmental clean-up is expected to facilitate their successful return or re-introduction. Habitat loss due to climate change may have substantial impacts if their breeding habitats are not ensured. Anticipated changes in salinity will have major impacts.

### How Climate Change Will Affect Wildlife

#### *Survival and Growth of Organisms*

Tidal marsh birds and mammals are particularly susceptible to the effects of inundation and storm events. Thorne et al. (2013) report that, during a strong winter storm, foraging habitat for black rails at a tidal

marsh was unavailable for many hours over several days (Table 5.2). The presumed stress on this species was not just due to the extreme level of water experienced by terrestrial animals, but also due to the duration of inundation, during which time black rails could not adequately forage in the marsh.

Inundation of marsh habitat can lead to direct mortality for terrestrial organisms that are poor swimmers, such as salt marsh harvest mice. Of particular concern, even for mobile organisms such as birds, inundation leads to greater risk of predation. Marsh inundation may force individuals into the upland edge of the marsh (i.e., Transition Zone), or to cling to tall vegetation, where individuals are much more susceptible to predation (Evens and Page 1986, Shellhammer 2000). Thus, marsh inundation interacts with predation (see below, **Species Interactions**). This mortality pressure, due to the simultaneous effects of marsh inundation and predation, is expected to increase due to climate change (Takekawa et al. in press). This mortality pressure is well demonstrated in the Ridgway’s rail. Overton et al. (2014) found that weekly survival of radio-marked Ridgway’s rails was negatively correlated with the extreme tide for the week. An example of especially high tides in a tidal marsh is depicted in Figure 5.3. The researchers concluded that predation was the predominant cause of the increase in mortality, and such mortality has likely affected recent population trends for this species (Nur et al. 2012).

**Table 5.2.** Inundation of marsh habitat during winter and early-spring storms. Percent of vertical vegetative habitat inundated during a winter storm in 2010 and an early spring storm in 2011 (from Thorne et al. 2013). The March 2011 storm had 80 -90% of the available habitat inundated and therefore functionally unavailable for wildlife. March 2011 storm had over 90% of the vegetative habitat under water during the Max SLH at all sites. Mean higher high water (MHHW) and maximum sea level height (SLH) were determined from water level loggers deployed in 2<sup>nd</sup> order channels.

	January 2010		March 2011			
	MHHW Non-Storm	MHHW Storm	Max SLH Storm	MHHW Non-Storm	MHHW Storm	Max SLH Storm
Coon Island	40.88	55.95	65.41	7.46	80.94	93.59
Petaluma Marsh	46.58	73.90	78.52	15.55	92.85	97.78
San Pablo Bay NWR	54.27	65.46	72.23	23.45	90.00	95.85

With longer inundation periods, aquatic species of higher salinity marshes, such as longjaw mudsuckers and Dungeness crabs, may have reduced exposure to avian predators and increased foraging times. However, higher temperatures during the periods of exposure may override the benefits of increased inundation. Neither the available data nor current climate models are adequate to evaluate these trade-offs.

Extreme temperatures may result in direct mortality. Increased mortality of lizards has been linked to an increase in ambient temperature as a result of climate change, because it reduces the number of hours per day that lizards can forage (Sinervo et al. 2010, 2011). While those studies were of terrestrial species, the same considerations may apply to estuarine species: that is, increased temperatures may result in energetic imbalance for some species, leading to increased mortality. However, lower flows and higher salinities appear to be better for growth of salmon in baylands (MacFarlane 2010). Thus, climate change impacts may provide a benefit to salmon in the estuary. However, of much greater concern are the likely thermal stress and dewatering effects upstream, particularly on steelhead. Increased temperature and increased CO<sub>2</sub> levels due to climate change will affect plant growth and survival, affect recruitment and dispersal, change competitive advantages among plants (Parker et al. 2012b; Grewell et al. 2014) and will change ecosystem level processes such as decomposition, nutrient cycling, organic accretion and food web support (Grewell et al. 2014).



**Figure 5.3.** Tidal inundation of habitat used by tidal marsh birds and mammals, such as the California Ridgway's rail, reduces refuge availability and increases susceptibility of nests to flooding. (Photo courtesy C. Overton, USGS).

The frequency of hypersalinity events in salt marshes (i.e., drought years when channel water salinity and soil porewater salinity along marsh creek banks rises well above marine salinity) will likely be associated with years of mass gumplankton dieback and reduced plant size (less cover, shorter gumplankton canopy), as is currently the case and was the case in the early 1990s (and 1970s-80s). At the same time, high salinity may favor pickleweed (*Sarcocornia pacifica*) over the short term.

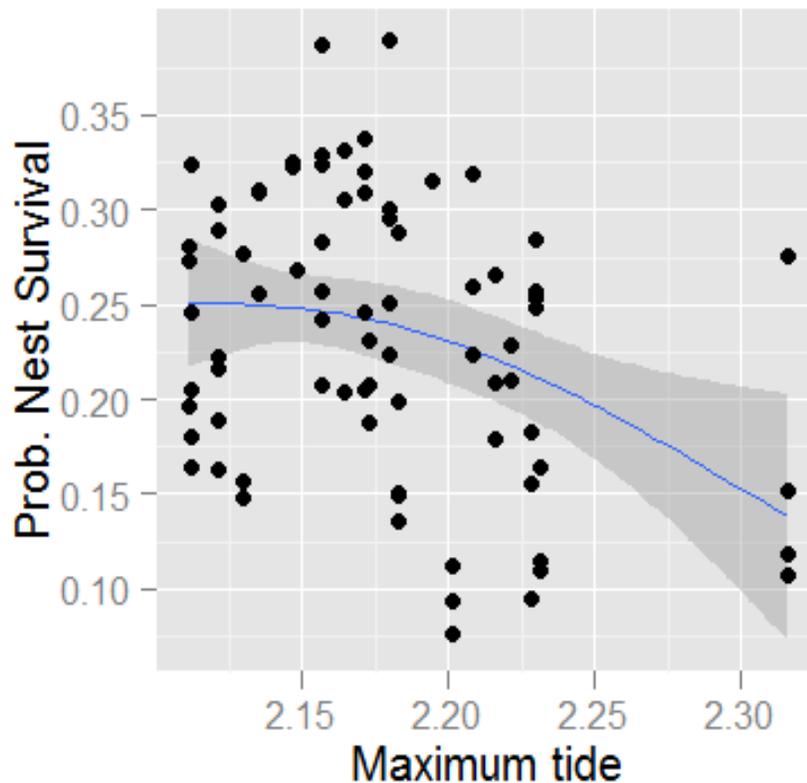
As inundation periods increase with sea level rise, the distribution and abundance of submerged aquatic plant species adapted to deeper flooding are expected to increase (Grewell et al. 2014). For many brackish plant species it is the combination of inundation and salinity that pose limitations when physiological tolerances are exceeded. Even plant species associated with salt marsh demonstrate high mortality when water levels and salinity are high (Parker et al. 2012a, 2012b; Woo and Takekawa 2012); the exception to this is *Spartina* (both native and non-native).

Shorebirds and diving ducks are energetically limited during the winter, as well as during their migration (Baldwin and Lovvorn 1994). Loss of mudflat habitat, which may result from sea-level rise exceeding rates of accretion of mudflats, could result in reduced survival of shorebirds (Crooks 2004, Galbraith et al. 2002, Galbraith et al. 2014). Furthermore, changes in water depth can affect accessibility of feeding areas for shorebirds. For diving ducks, they are more likely limited by the availability of prey species directly; climate-change induced change in prey species composition or abundance may lead to reduced survival.

Additionally, invasion or establishment of non-native species may lower survival as a result of competition, predation, or habitat change (see below).

### Reproductive Success

Tidal marsh birds and other wetland-breeding birds (e.g., shorebirds) are susceptible to reproductive failure due to flooding of nests (Powell et al. 2002, Greenberg et al. 2006, Nur et al. 2012). Increased risk of nest-flooding reflects the combined effect of sea-level rise due to global warming, El Niño events, and storms, are all likely to increase due to climate change, as discussed above. Both the changes in mean high water levels and an increased risk of extreme events (storms, floods) are a concern. In the song sparrow, especially high water levels, whether due to extreme tides or storms, are strongly correlated with nest failure (Figure 5.4; Nur et al. 2012). Extreme water levels affect a nesting attempt that is underway. However, timing of extreme water levels is also important: the complete nesting cycle requires about 25 days. Whereas an extreme tide can cause failure of a nesting attempt that is underway, song sparrows will re-nest soon after the failure. However, if another extreme water event occurs within 25 days, then the re-nesting will also fail, which may lead to reproductive failure for an entire breeding season. A population model indicated that increased water levels, resulting from climate change, could, in the future, cause populations of tidal marsh song sparrows to decline by more than 80% over a 50-year period (Nur et al. 2012, Takekawa et al. in press). Black rails and Ridgway's rails exhibit the same sensitivity to nest failure as a result of extreme water levels. Nur et al. (2012) and Overton et al. (2014) inferred that nest failure due to flooding was a concern for Ridgway's rails (documented by Schwarzbach et al. 2006) and could reduce future population viability (Takekawa et al. in press). Small tidal marsh mammals (salt marsh harvest mice, bayland shrews) may also be at risk of reduced reproductive success or lower offspring survival due to flooding.



**Figure 5.4.** Song Sparrows: Nest survival and extreme tide. The greater the maximum tide experienced during the course of an entire breeding season, the lower is probability a nesting attempt survives to fledging (Nur et al. 2012). Extreme tides are expected to increase substantially in the future, especially over the next 30 to 100 years.

Amphibians, including the California toad and California red-legged frog, require freshwater ponds of sufficient depth and temperature. In particular, breeding ponds need to maintain appropriate conditions long enough for offspring to mature. Climate change may result in ponds drying out too fast or with excess salinity.

Many aquatic species have historically shown better survival or reproduction in years of higher outflow. The predicted restriction of outflow events to early in the water year is likely to restrict the spawning success of both longfin smelt and delta smelt. Particularly for delta smelt, the prolonged summertime conditions are likely to reduce survival and fecundity as the salinities they occupy in the summer and fall move upstream into less productive conditions of the delta. The broader distribution of longfin smelt may allow them to be less affected due to their ability to retreat to more oceanic waters. Pacific herring require a combination of solid substrates and appropriate salinity that is apt to become disconnected under the future climate. Suitable salinities will move upstream into San Pablo Bay, where the appropriate solid substrates are rarer, thus, eggs will be deposited on inappropriate substrates or in even greater densities on the limited patches of appropriate substrate. In either case egg survival is likely to decrease. Effects on the young, which use a wide variety of habitats, may be less severe.

Harbor seals require tidal flats of particular characteristics to haul out and birth pups (Figure 5.5). Loss of adequate haul outs (as a result of erosion or flooding of current tidal flats) is a concern, resulting in reduced reproductive success. Similarly, shorebirds require suitable breeding locations, including beaches and mudflats, which may be lost due to sea-level rise.



**Figure 5.5.** Harbor seals depend on tidal flats to successfully rear pups. Sea-level rise will threaten current haul-out sites. Photo courtesy of S. Allen.

Many important plant species of the high-mid marsh are sensitive to salinity extremes. Many of these species may depend on a low-salinity period for germination during the winter and spring. This is of greatest concern for uncommon local endemic species, such as Suisun thistle and water hemlock (*Cicuta maculata*), which are patchily distributed, but applies to other plant species as well. For Suisun thistle, in addition to effects of salinity, inundation stress due to sea-level rise will also reduce the first order tidal channel habitat supporting Suisun thistle, and thus limit the distribution and abundance of populations throughout Suisun Marsh (Fiedler et al. 2014).

## Species-Interactions

### Predation

Predation is of high concern for animal species in bayland habitat (Greenberg et al. 2006, Nur et al. 2012). Tidal marsh-inhabitants are particularly susceptible, often subject to predators from adjoining uplands, including developed land adjacent to baylands (see Science Foundation Chapter 4). However, it is not clear that predation levels will, in general, increase due to climate change. In any case, current levels of predation are already straining the resilience of many bayland populations, leading to declining populations or ones that are not able to recover (Takekawa et al. 2012).

We suggest two pathways that may lead to increased predation on terrestrial species.

- (1) Greater access by, and exposure to, non-native and human-associated predators. As a result of climate change, directly and indirectly, some predators (including those that are associated with or benefit from association with humans) may be more prevalent than at present or exposure to such predators may increase. Climate change may lead to “prey-switching,” which can lead to heightened predation rates. Human population increase can contribute to this pathway for human-associated predators.
- (2) Increased predation mortality specifically due to limited refugia, especially at times of stress or sensitive life stages; this is of concern especially regarding high water events.

These two factors interact, as shrinking wildlife habitats about human habitats and other sources of predation. Habitat is shrinking for species such as American avocets and snowy plovers not only due to climate change, but also due to extensive restoration activities impacting their current nesting habitat.

Management actions can affect one or the other of these two pathways, depending on the species and habitat under consideration. With regard to predator access, specific predictions are hard to make, other than that climate change will allow some predators to invade or establish themselves, which otherwise would not. Currently, California gulls are a species that has great predation impact on desirable bird species of bayland habitat (Shuford 2008). We are not able to predict whether climate change will increase predation rates, but that possibility is a concern, especially because habitat restriction (e.g., nesting habitat of snowy plovers) is increasing the susceptibility to predation.

Access of predators is often linked with human actions. Stocking of mosquitofish or sportfish into temporary ponds can have large impacts on reproduction of California red-legged frogs and California toads. Increased inundation and higher sea level is likely to produce mosquito problems in areas that at present are rarely wet enough. Levee enhancement and other efforts to buffer human infrastructure from the impacts of climate change are likely to improve access for predators, especially human-associated predators like raccoons, cats, and rats (Takekawa et al. 2006b).

High water events make tidal marsh birds and mammals especially vulnerable to predation. This is especially of concern for rail species and the salt marsh harvest mouse (Overton et al. 2014; Shellhammer 2000). River otters require dense vegetation refugia at times of high river flows. Designing habitats that function as refugia under extreme conditions will be an important part of planning for the impacts of climate change.

### *Disease*

Risks due to disease are expected to increase (Harvell et al. 2002). Harvell et al. (2002) conclude: “[M]ost host-parasite systems are predicted to experience more frequent or severe disease impacts with warming” of the climate. In the temperate zone, shorter, milder winters are expected to increase disease spread (Harvell et al. 2002). Avian cholera, particularly on waterfowl, is currently a concern in the San Francisco Estuary (Takekawa et al. 2006b). However, projected climate change impacts for the incidence of the disease are not available. River otters are subject to disease, including emergent diseases; climate change resulting in reduced prey may affect susceptibility of otters to disease. Harbor seals may be subject to pathogen shifts as a result of climate change, especially through increased proximity to terrestrial carriers of morbillivirus (dogs, cats, raccoons, skunks), *Leptospira* (rats), *Toxoplasma* (felines) and *Sarcocystis* (opossums) (Greig et al. 2014).

Amphibian chytridiomycosis (caused by *Batrachochytrium dendrobatidis*) is of great concern for the California toad. Some warming may cause *B. dendrobatidis* to spread or increase, but substantial increase in temperature may actually reduce the pathogen. As with many climate change impacts, this is of potential concern, but it is difficult to assess the risk due to climate change.

Incidence of plant diseases will likely be affected by many factors as a result of climate change (Garrett et al. 2006). Strong plant-microbial linkages, including that of mycorrhizal fungi, may help reduce disease (de Vries and Bardgett 2012); however, climate change predictions regarding microbes are difficult to obtain. For both plants and animals, there is concern that pathogens will evolve faster in response to climate change than host populations, and therefore spread more quickly with more virulent results (Garrett et al. 2006).

### *Community Composition*

Species distribution models (Stralberg et al. 2009a, Ackerly et al. 2012) indicate that climate change will create new assemblages of species, thus changing the nature of competition among species. Climate change may cause non-native species to invade the baylands as well as species native to warmer parts of California to move into the San Francisco Estuary. Conversely current species may move out of the Estuary, as has been discussed above. Such changes in distribution could have direct and indirect effects. For example, a change in a predator could alter the balance among several competing species; a potential competitor may be “released” from limitations due to a reduction in predation. In general, the effects of these new assemblages on target species are not known, but there is potential for reduction in population viability. More studies are needed to identify the pathways by which a change in community composition affects target species. Because the risks are not yet identified, establishing a surveillance-monitoring program is recommended.

Interactions among aquatic species can often be affected more by changes in characteristics of water (salinity, temperature, dissolved oxygen, etc.) than by changes in the physical substrate that is the principal focus for terrestrial species. For example, re-establishment of oyster beds in San Francisco Bay is desired by many and may be an important tool in reducing the impacts of storm surge. Reduced sediment loads associated with future climate change impacts may facilitate the establishment of oyster beds. However, warmer temperatures and reduced circulation combine to produce lower dissolved oxygen levels that promote the spread of oyster drills that can decimate the population. As another example, the introduction of various Asian gobies that prey and compete with tidewater gobies may preclude the reestablishment of tidewater gobies into all of their former areas. Management actions that make such areas less suited for the invaders may be a necessary first step to re-establishing the native species.

## Habitat Structure and Change

### *Plant Species Diversity and Structure*

Species diversity is an important component of ecosystem resilience. Maintenance of plant species diversity, in particular, may be key: high plant diversity increases disease resistance (deVries and Bardgett 2012). Plant species diversity in a California tidal marsh was severely reduced by extreme events over the course of several decades, but the loss of diversity was particularly acute in 1984 due to the sequence of extreme events, in this case river mouth closure followed by drought (Zedler 2010).

Plants play an important role in providing suitable habitat, foraging locations, breeding sites and substrate, concealment from predators, etc., for many bayland animals. Vegetation structure is of particular importance for tidal marsh birds and mammals (Shellhammer 2000, Spautz et al. 2006). Gumplant (*Grindelia stricta*) plays an especially important role in tidal marsh because the uppermost part of this shrub can remain above high tides. However, gumplant is subject to die-backs, and thus mortality of this species (due to hydrology or salinity pressures) is a concern. Bulrush and tule play an important structural role in Suisun Marsh; increased salinity will cause important habitat change for tidal-marsh birds and mammals.

The important and complex role played by plants in structuring habitat and influencing wildlife species is well-illustrated by the non-native smooth cordgrass, *Spartina alterniflora* and its subsequent hybridization with the native Pacific cordgrass, *S. foliosa* (Guntenspergen and Nordby 2006, Grosholz et al. 2009). The hybrid *Spartina* has caused habitat changes at lower elevations in or adjacent to the marsh, leading to conversion of mudflats into vegetated, low marsh; at intertidal elevations native plants have been displaced by the invasive hybrid. Nevertheless, the invasive hybrid appears to have had a short-term beneficial impact on the Ridgway's rail. Lack of cover and refugia from high tides (during the winter and breeding season) have contributed to population declines of the Ridgway's rail prior to the invasion by *S. alterniflora*. In marshes which lacked adequate cover and/or refugia, the invasive *Spartina* provided needed vegetation cover.

However, there are numerous reasons to be concerned about changes in the habitat structure and characteristics as a result of establishment of the invasive *Spartina*, not the least of which is loss of critical mudflat habitat for shorebirds to forage in (Stralberg et al. 2004). Loss of the native *S. foliosa* is a second concern. The invertebrate assemblage of species is also altered as a result of the invasion (Grosholz et al. 2009). Finally, nest survival was reduced for tidal marsh song sparrows breeding in marshes which were invaded by hybrid *Spartina* (Nordby et al. 2009).

### *Habitat Connectivity*

Compared to pre-human settlement conditions, bayland habitat is much more fragmented (SFEI EcoAtlas; Science Summary, this Report). Thus, connectivity among habitat patches has been reduced. Our emphasis in this chapter is on “functional connectivity,” such that organisms, plants and animals, can move from one habitat patch to another, rather than contiguity of habitat. Even if connectivity is not further reduced in the future, the loss of connectivity, compared to pre-settlement, represents a serious risk to wildlife, because low connectivity reduces the resilience of wildlife populations.

Habitat connectivity is especially important in two regards: 1) For populations to be robust and resilient, successful dispersal is an important requirement. An isolated population is neither robust nor resilient: it is only a matter of time before it goes extinct. However subpopulations connected by dispersal are much more likely to persist. Habitat connectivity allows such a metapopulation to persist despite unpredictable and, in some cases, unfavorable conditions, for certain time periods or at specific locations (Hanski 1994). Migrant species also benefit from connectivity but their requirements will differ from that of year-round resident species. 2) Under several future scenarios, current bayland habitat (tidal flats, tidal marsh, etc.) is expected to change. What is currently high tidal marsh may become low marsh, low marsh may become tidal flats, tidal flats may become subtidal, etc. There may or may not be a net loss of habitat, but what we can expect is marsh (and other habitat) transgression, i.e., marshes will move, to the extent possible (Science Foundation Chapter 1). The implication is that habitat patches that are currently suitable will no longer be suitable, but that other areas within the landscape will become suitable habitat. As a result, if species (plant and animal) are to persist in the San Francisco Estuary, they will need to be able to move. And that will require connectivity of suitable habitat.

The ability to disperse successfully is an especially important limiting factor for many plant species. High tidal marsh annual forbs include several rare or endangered species (e.g., *Chloropyron maritimum*, *C. molle*, *Castilleja ambigua*). These species have limited ability to spread or recolonize (or even maintain their number), and recruitment is also limited due to competition with non-native species (Grewell et al. 2013). As with other non-native invasive species, there are a number of annuals that are strong colonizers. Thus for the native marsh species of concern, active translocation may be required. A similar situation exists for plant species of vernal pools.

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## **BACKGROUND TO RECOMMENDED ACTIONS**

### *Strategy for Resilience*

Several themes emerge from synthesizing the 32 case studies regarding impacts of climate change and how these can be addressed. These case studies and recommendations are based on the monitoring and special studies of wildlife in the estuary, despite the data gaps we describe. Climate change will cause habitats to shift, species to move, and communities to reorganize; our current monitoring programs will have to adapt if they are to provide the information that future management and protection will depend on. Developing a modeling framework for our changing estuary will require substantial effort and is essential to the management recommendations.

## **(1) Ensure Suitable Habitat in the Future**

Habitat for bayland species will be difficult to protect because of the uncertainty associated with future projections (Stralberg et al. 2011, Veloz et al. 2013; Science Foundation Chapter 2). The uncertainty of climate projections and sea-level rise are of global concern and intensive predictive efforts are underway. For baylands in the San Francisco Estuary there is also considerable uncertainty about future suspended sediment loads and distributions, as described in Science Foundation Chapter 1. Two of the habitats of great concern for wildlife are tidal marsh habitat (both saline and brackish) and tidal flats. The range of possible amounts and distribution of tidal marsh has been modeled by many, but less so for tidal flats. However, tidal flats are a key resource for many wildlife species, such as shorebirds (for foraging) and harbor seals (for hauling out to rest and nurse their pups). In addition, estuarine beaches are an important habitat type for some plants (as described in the case study), shorebirds, and potentially grunion.

Concern about climate change is likely to lead to armoring levees, e.g., creating higher, more substantial seawalls. Armoring may protect humans and human infrastructure but will largely eliminate shallow habitats on the bayward side of the levee. As a result, important habitats, especially tidal marsh, tidal flats, and estuarine beaches will likely be reduced. An alternative to this armoring to protect humans and human infrastructure is for marshes, mudflats, beaches and oyster beds to play an important role in dissipating tidal energy from the levee and redirecting that energy onto nearby, less armored parts of the bay margin. In this paradigm, natural habitats provide not only ecosystem services to wildlife and plants by creation and protection of habitat, but also provide protection of levees and seawalls that are placed upslope of these natural habitats.

Some habitat types depend entirely on substantial human intervention: managed wetlands including commercial or former salt ponds, diked marshes, and muted tidal marshes. These habitat types, though dependent on continual management input, have a role to play with regard to the future habitat mosaic of the San Francisco Estuary; emphasis cannot remain solely with “self-maintaining habitat,” to use the Goals Project (1999) label. Of particularly value for wildlife are **managed ponds**, a category that includes several types of ponds (salt ponds, storage/treatment ponds, etc.) that are managed to support wildlife. While no species in the San Francisco Bay region is a managed pond obligate, the loss and degradation of natural habitat for waterbirds in the past century has led to strong reliance on managed ponds at present. The Goals Project in 1999 set acreage goals for managed ponds, which entailed substantial recommended reduction in the amount of managed pond habitat, coupled with management directed at maintaining conditions (with regard to salinity and depth) that will support high densities of shorebirds, waterfowl, and other waterbirds, despite reduced acreage. Since publication of the Goals Report in 1999, we have acquired much specific information that will be invaluable in managing this habitat so as to maintain or increase abundance of shorebirds and waterfowl (Warnock et al. 2002; Takekawa et al. 2006a, Takekawa et al. 2009 ; Brand et al. 2014); see Theme #2, below (**Provide necessary habitat features**).

We highlight the importance of providing sufficient habitat specifically with regard to managed ponds for two reasons. Climate change will lead to increased water levels and changes in salinity (Science Foundation Chapters 2, 3 and 4) putting additional stresses on the network of levees and water control structures necessary for maintaining optimal managed pond conditions. Second, restoration of tidal marsh through the conversion of managed ponds has resulted and will result in less acreage available. Thus, there will be important challenges in future years to maintaining managed ponds so as to reduce adverse impacts on waterbird populations using this habitat. For example, investment in new water control structures may be required (Science Foundation Chapter 2). Increased reliance on pumping may also be necessary. However,

design and management of these ponds should not be pursued in isolation: natural habitat, including tidal marsh and tidal flats provides important resources as well. In any case, it is clear that future environmental conditions such as higher water surface elevations, frequency and intensity of storm events, and regional salinity shifts will make it difficult to maintain target habitat conditions inside the ponds. For some current managed ponds, it will be difficult to retain those ponds as such and thus relocation (taking into account sea-level rise and future hydrological conditions) may be necessary. Thus, investment must consider both the long-term future of managed ponds as well as the future restoration of tidal wetlands containing ponded areas and other features that benefit the suite of species currently relying on managed ponds.

Diked marshes represent another important habitat; they can provide important resources for species usually associated with fully tidal marshland, such as salt marsh harvest mice and the two bayland shrew species. Case study authors recommend that diked marshes be included in the future mix for these species. Tidal marsh birds (such as song sparrows) may also benefit from muted tidal marshes; their value should be explored.

## **(2) Provide necessary habitat features, locally and across the landscape**

Characteristics of a given habitat, at the local and landscape scales, are as important as habitat quantity. One important habitat feature is the width and extent of the marsh/terrestrial transition zone, which is treated in detail in Science Foundation Chapter 4. Salt marsh harvest mice and bayland shrews need a broad, gently-sloped natural transition zone with substantial vegetation; such a zone likely benefits black rails as well as other tidal marsh birds (Evens and Page 1986). Several plant species benefit from such transition zones. The transition zone has been degraded or lost (Science Foundation Chapter 4); this zone is narrower than in the past, and now abuts artificial levees and urbanized, or otherwise developed, lands. Restoration or re-creation of natural transition zones (as well as the bayland and terrestrial habitats on either side of them) will help many wildlife species. The need is acute especially as new levees are built to protect human infrastructure from climate change impacts. It is equally important to focus on the nature of the terrestrial habitat that borders the marsh habitat: predators and invasive species often enter the transition zone on the terrestrial side. Upland areas that will allow marsh transgression and transition zone values are likely to become rare, so all opportunities should be considered.

Refuge habitat from predation and extreme water levels (e.g., during especially high tides) is already of high importance and will become more important to wildlife populations in the face of climate change. Refugia may also be needed with respect to drought, which not only leads to drying up of pools and ponds, but also can lead to hypersalinity. Drought, hypersalinity, and extreme water levels are all likely climate change impacts; predation rates may also be affected. In particular, climate change may provide avenues for invasion by non-native or “nearby-native” species to move in. Human-associated predators (e.g., crows and ravens, gulls) already pose substantial risks which need to be addressed or in the future these risks may increase.

Ancient tidal marshes are characterized by a highly dendritic, sinuous network of tidal channels (Goals Project 1999). Tidal marsh-dependent plants and animals benefit from such channels. Thus, restoration efforts should give this habitat feature precedence. In this way, a restored marsh may be more valuable to wildlife than a 150-year old marsh created during the gold-rush period. In the face of climate change, marshes and adjacent habitat can be designed and placed to provide more resilience to wildlife than current older, or even ancient marshes do.

Design considerations can allow smaller areas to do more for their dependent populations.

For example, topographic relief in a marsh is of value; enhancing that topographic complexity through creation of marsh mounds and berms can enhance marsh heterogeneity, increase plant species diversity, and provide some barriers to water flow or refuge from highwater events.

Another example is to build gently-sloped transition zones, fed by brackish flow (the “horizontal levee”; TBI 2013). Freshwater ponds (for amphibians), vernal pools, and saline ponds can all be maintained with salinity and water levels subject to control to benefit wildlife.

Salinity is an important habitat characteristic for many species as discussed in this chapter. In particular, salinities have generally been lowered in managed ponds both in the North Bay and South Bay, as part of a long-term management strategy to ultimately convert managed former salt ponds into tidal marsh habitat (Stralberg et al. 2006), with fewer ponds exhibiting extreme salinity. The result has been a substantial increase in diving and dabbling ducks (Pitkin and Wood 2011). But such change has not necessarily been as favorable for shorebirds, some of which, such as the western sandpiper and American avocet, rely on the high densities of invertebrates (e.g., brine flies) found in hypersaline ponds (Brand et al. 2014). At the same time reduction in depth of water in managed ponds increases accessibility of foraging habitat for shorebirds (Warnock et al. 2002, Brand et al. 2014). Recent reduction in depth in some managed ponds has led to observed increases in shorebird numbers (Pitkin and Wood 2011). Thus, maintaining the optimal balance of salinity and water-depth in managed habitat will be important in the near future and for some time to come.

Management of salinity and water depth is an example of a recent trend, which will continue and accelerate in the future, to emphasize (from a management perspective) on quality of habitat, both with regard to features within a marsh or pond and also with regard to configuration of habitat. In this way, for example, a reduction in acreage of managed ponds can still potentially result in increases in abundance of shorebirds and diving ducks (Pitkin and Wood 2011), provided there is the ability to provide the specifics of salinity and water depth that benefit a range of waterbird species. Wildlife will require comprehensive planning efforts, such as a scenario planning effort for the South Bay Salt Pond Restoration Project which has recently been completed.

### **(3) Maintain or augment population resilience by addressing other stressors**

A resilient population is better able to tolerate the effects of climate change, especially stresses due to extreme events such as droughts and floods. Such climate change impacts are difficult to address directly, but further reduction of known stressors will add to a population’s ability to withstand new stressors (and is addressed in Chapter 2 of the main report – *New Opportunities: How We Can Achieve Healthy, Resilient Baylands*). A resilient population has sufficient reproductive success and survival to offset mortality, including occasional “catastrophic” mortality. Hence, knowledge of reproductive and survival rates is extremely desirable, though difficult to measure in the field. As a result, the monitoring of trends in abundance provides an initial and cost-effective means to assess if a species is in trouble. Where abundance has declined over time, research and management need to respond quickly to reverse that trend. Low nest survival of tidal marsh song sparrows has been implicated in their recent declines throughout the San Francisco Estuary (Nur et al. 2012). Ridgway’s rails have demonstrated an increase in number relative to the 1990’s, but a decrease from 2007 to 2013, with low first-year and adult survival the prime contributor (Overton et al. 2014). Such studies provide indication of where management should focus on to augment resilience.

Stable abundances or short-term positive trends provide no guarantee of the long-term health of a population. Therefore, a prudent approach is to address mortality sources for all target species, whether or not they are declining.

The following are stressors whose impacts could be reduced, independent of climate change, thus increasing population health and supporting resilience. Specific actions are discussed in Chapter 2 of the main report – *New Opportunities: How We Can Achieve Healthy, Resilient Baylands*

- **Predation, affecting both survival and reproductive success.** Section III summarizes these effects.
- **Contaminants.** Mercury contamination is of particular concern to birds and mammals. Pyrethroids are a major concern for aquatic species. “Emerging” contaminants (SFEI 2013 [RMP]) have been implicated in reducing the survival and reproductive success of many species. There are many other contaminants of concern.
- **Nuisance and invasive species; disturbance.** Management that targets nuisance and invasive species is often less controversial than other actions. Disturbance by humans (e.g., due to recreational use; dogs and their owners) can be reduced. Shorebirds and waterfowl benefit from reduced disturbance, as do harbor seals. If grunion return to the estuary it will be important to protect their spawned eggs from off-road vehicles.
- **Disease susceptibility.** Susceptibility can be reduced through improvement of physical condition, which in some cases reflects prey availability. Harbor seals, frogs and toads, and colonial (flocking) waterbirds are example of species subject to disease.

#### **(4) Maintain resilience by increasing recruitment and dispersal success and facilitating movement at the local scale**

Successful recruitment of offspring into the adult population is a necessary requirement for viable populations, and is even more important to produce populations robust to climate change and other stressors. The fragmented nature of current baylands places a strain on species’ variability. The current patch of habitat occupied by a breeding animal or plant may be appropriate, but what is nearby may be inappropriate habitat or of poor quality. And yet, dispersal ability is limited for many bayland species of concern. Longjaw mudsucker, tidewater goby, vernal pool plants and invertebrates, salt marsh harvest mouse, and the bayland shrews appear to have limited dispersal abilities (Table 2). Rail species (black rail and Ridgway’s rail) demonstrate low dispersal rates even when such movement would be adaptive (Overton et al. unpublished). Evolution of high site fidelity, and therefore low dispersal ability by these species, may have been adaptive in the past. In the current, fragmented landscape it is a problem. Changing landscapes due to climate change will surely worsen conditions for less dispersive species (see theme #5, below, **Manage for uncertainty**).

Restoration designs can address the problem of fragmentation by targeting functional connections (i.e., allowing effective movement and dispersal). Highways, levees, and other structures can be important impediments to successful dispersal; steps must be taken to address these barriers.

Species that live along water courses or the bay margin are better able to disperse because their habitats are physically linked and they exhibit good dispersal abilities. Such species include river otter, salmon,

shorebirds, and harbor seals. Often these species can move within a water course or along the bay margin as conditions in one area change and they disperse easily to new areas because their habitats are connected by water. For these species, it is important to ensure that there are no barriers to successful movements among suitable habitats and that all habitats are within the dispersal distance of a healthy source population. Such barriers would include dams or channelized sections of streams or long unbroken stretches of levees on the bay margin.

Small population sizes increase population vulnerability. Small populations are more prone to local extinction, due to unpredictable mortality events and reduced fecundity reproduction when density is low (the “Allee” effect). Ridgway’s rails may already demonstrate this phenomenon. The dangers of small population size are accentuated when sub-populations are isolated from each other. Small populations may require translocation from other locations or other active management to boost population size.

### **(5) Maintain resilience by managing for dynamic landscapes**

The landscape of the San Francisco Estuary will change. The nature of that change is not well-established, though we expect that the velocity of climate change, and therefore its impacts, will be high in the human-altered landscape (Loarie et al. 2008, Ackerly et al. 2012). Not only will there be somewhat predictable change (sea-level rise will accelerate, temperatures will increase, including duration and frequency of heat spells, etc.), there will also be change of an unpredictable nature. Thus, as discussed above, wildlife populations will be subjected to more frequent, more extreme, and more unusual, stresses from California’s changed climate. More habitat (see #1, above), possessing the appropriate characteristics (see #2, above) and allowing for dispersal and movement among suitable habitat patches (whether currently occupied or not; see #4, above) will all help species cope with climate change. But the changing and unpredictable nature of future landscapes will require even greater effort, monitoring, and specific planning. Thus, restoration design needs to go beyond simply improving on the present (more, better quality, better connected habitat). Therefore one recommendation (see Chapter 2 of the main report – *New Opportunities: How We Can Achieve Healthy, Resilient Baylands*) is to anticipate where mudflats or tidal marsh may migrate in the future, and design restoration or habitat enhancement accordingly.

To persist, plants and animals will need to move into suitable new habitats (Grewell et al. 2013). Such movement is required because once-suitable habitat will become unsuitable and vice versa. We anticipate that local occupied patches may be extirpated, or reduced to very low levels, due to the combination of extreme events and other stressors (disturbance, contaminants, non-native species, etc.). Protection of wildlife species will require designing landscapes with dispersal needs in mind. For some species of limited dispersal ability, or for which current barriers are too high, active translocation of individuals will be required, as currently occupied habitats are lost or degraded and new habitat is produced in other areas.

An additional protection against extinction is maintaining **genetic diversity**. The currently diverse and patchy landscape requires genetic diversity, and with future (unpredictable) change, genetic diversity is even more important. Many species of concern in the Baylands are composed of genetically distinct populations, and in some cases distinct subspecies; such differentiated species/subspecies include song sparrow, California red-legged frog, salt marsh harvest mouse, bayland shrews, black rail, and salmon. These species or subspecies represent valuable genetic diversity and adaptation to local conditions as evolutionary significant units (ESUs; Fraser & Bernatchez 2001). Often, however, these populations are small and so require high within-population genetic variation to maintain viability (Soulé 1986). We need to manage these genetically differentiated species to maintain their resilience and facilitate the re-colonization of suitable habitat following catastrophes. Re-colonization may occur by a different sub-

species or population than was originally present. This may be a natural aspect of rapid evolution brought on by the impacts of climate change. Maintaining spatially distributed and connected habitat for these species may be important in preserving the genetic diversity currently present so that bayland populations can respond to changed conditions on different time scales of climate change.

Isolated populations such as those of rare marsh plants and invertebrates of vernal pools represent unique products of evolution. However, these isolated populations have very little cross-breeding, and thus loss of a population due to catastrophe may represent a complete loss of some genetic diversity and must be avoided.

## **(6) Manage for uncertainty, including for extreme events and for the “unforeseen”**

Uncertainty is used to describe two very different concepts:

1. Our lack of knowledge or precision about the physical parameters of the environment and the biology of the various species.
  2. Unpredictable events that we know will happen but we don't know when, how much, etc. We can expect to be subject to unprecedented events, but cannot predict the specifics.
- Both kinds of uncertainty must be accommodated in developing management actions.

Difficulty in making predictions regarding climate change is discussed in Science Foundation Chapter 2. Changes in geomorphology and hydrology are also difficult to predict. Sedimentation rates are important parameters for many baylands that are likely to change, especially in light of future restoration projects and upstream water management. Future decisions regarding the maintenance or construction of infrastructure (e.g., levees) will affect future habitat. The result of all these influences is substantial uncertainty regarding future habitat extent and distribution. This source of uncertainty must be addressed in planning (Veloz et al. 2013).

However, in some cases, it is only the rate of change that is unknown. We know what will happen, but not when. Therefore one approach is to develop triggers for management action; when thresholds are crossed, management action is triggered. As described in the marsh plant case studies, examples of thresholds to be concerned with are:

- Recurrent failure of levees; overtopping or breaching of levees.
- Significant reconstruction or armoring of levees or new hydraulics.
- Low marsh vegetation dominates mid- to high-marsh in the marsh plain, where it wasn't before.
- Large-scale conversion of brackish marsh to saline marsh.

An additional component of uncertainty is the widespread lack of basic information for many species of concern. This is the case not only for rare species such as the bayland shrews, but for quite common species like river otters. River otters are becoming much more common in baylands, but we are ignorant as to whether that is the sign of a burgeoning population or of movement downstream from more disturbed areas upstream. In the case studies, important knowledge gaps for managing the species are described. The state-of-the-knowledge is particularly poor when it comes to what we need to know to manage for climate change. For many species, we don't know if populations currently are stable, declining, or increasing. For species that breed elsewhere, even if we have current trends for San Francisco Estuary, important

information may be missing. For example, trends in wintering waterfowl (surf scoters and scaup) may reflect causal influences acting on their breeding grounds.

The approach needed to develop and respond to uncertainty is **adaptive management**, and that approach is particularly needed when it comes to climate change. We recommend the development and application of **population models that can incorporate environmental variability**. Such models can:

- consider different future scenarios,
- identify important bottlenecks and thresholds for species of interest, and
- incorporate environmental variability especially of the stochastic kind, i.e., the unpredictable extreme events.
- Such models can evaluate sensitivity of output to uncertainty of both kinds: lack of knowledge and unpredictability.

Additional monitoring and research studies must target important data gaps, evaluate model assumptions, and validate models. The models can be used to evaluate resilience and explore how resilience can be increased. The long-term response of wildlife populations to extreme events can be modeled. Monitoring could be conducted immediately after an extreme event to determine the population consequences of the event. Further monitoring can evaluate the recovery of the population, and this can be compared to model predictions. Information on turn-over (extirpation and re-colonization) of sites, including newly available sites as a result of restoration or habitat change, is needed to construct models that project how populations will grow, shrink, spread, or disappear across the entire bayland landscape. As we may need a stockpile of sandbags to deal with a broken levee, we need a toolbox for wildlife management to address the expected heat waves, floods, droughts, storm surges, winds, etc.

### Important Data Gaps and Evaluation of Uncertainty

Here we summarize information on sources of uncertainty and important data gaps. Galbraith et al. (2014) provide a good outline of information needs for determining shorebird response to climate change and their work provides a good model for other species or groups. Uncertainty in climate change drivers is discussed in Science Foundation Chapter 1 and also in the section above (“Managing for uncertainty”).

#### 1. Information on population trends.

As previously noted, for many plant and animal species, we do not know if bayland populations at present are stable, declining, or increasing. This lack of information makes it difficult to develop appropriate management actions, as well as to prioritize those actions. Thus, an emphasis of trend monitoring, at the appropriate spatial scale is needed (see Section above).

One species of widespread conservation interest for which there is little information is the river otter *Lutra canadensis*. Otter are being seen more frequently and are reported from all bay area counties, including San Francisco. This may represent either a burgeoning population or a species losing all of its suitable habitat upstream. As an apex predator, the otter has significant ecological impacts on its prey species and may be a sentinel species for a variety of stressors that multiply through the food web. But at present little of its status or needs or impacts are understood.

For many bayland mammals, herps, invertebrates, and plants, we know little about their population trends. In contrast, for fish, population trends are known, at least in the open bay, as a result of surveys by the California Department of Fish and Wildlife (SEIT 2011), though not necessarily in bayland habitat. In addition, some individual fish species have been well-studied.

Population trends for many bird species in the San Francisco Estuary are known (Pitkin and Wood 2011). However, a number of bird species in the estuary breed elsewhere; for example, diving ducks are mainly present during the winter period. Use of the San Francisco Estuary is very important to the ducks, but winter-time trends in population are hard to interpret. Do the trends reflect population dynamics on the breeding grounds, or do they reflect shifts in their distribution? A shift may simply reflect increased or decreased suitability elsewhere rather than reflecting conditions or management action in the San Francisco Estuary. A similar problem exists for trends of western sandpiper and other shorebird species.

## **2. Information on underlying vital rates: survival, reproductive success, recruitment rates.**

For many species for which population trend estimation is possible, we have little or no information on adult survival, reproductive success, juvenile survival, or recruitment success. In some cases, we may have information on just one of these rates; for example, reproductive success for tidal marsh song sparrows and least terns in San Francisco Estuary has been studied for a number of years as is the case for several dabbling duck species in Suisun Marsh. In the case of Ridgway's rails, there is some information on juvenile and adult survival, but very little current information on reproductive success. For California red-legged frogs, information on adult survival is lacking, but studies indicate that egg survival and larval survival are very low, pointing to these two parameters as key with respect to population viability and resilience and thus important targets for management action. Thus, information on even one demographic parameter is extremely valuable.

## **3. Causal influences on variation in demographic parameters.**

In addition to estimation of demographic parameters, e.g., obtaining mean values, it is also essential to characterize variation (across space and time) in demographic rates, especially in relation to environmental variables. If we can understand how current populations respond to environmental variation, we will be better able to project how wildlife will respond to climate change and other stressors. In short, lack of information on demographic rates and how they are influenced by the environment represents a key uncertainty and a data gap that needs to be addressed.

The nature of variation in demographic rates is poorly known: How plastic are these rates? How do they respond to environmental variation, and over what spatial scale? One of the most important questions is, To what extent do these rates respond to management action? Empirical studies of birds indicate that a small difference in adult survival rates has a large effect on population trend. However, often reproductive success is much more responsive to management actions than is adult survival in many cases. Such information is crucial to projecting impacts of climate change on population health of wildlife and developing management actions to address these impacts (Nur et al. 2012).

How will changes in the habitat due to invasion by non-natives (e.g., hybrid *Spartina*) affect survival and reproductive success, both in the short-term (which may be feasible to study) and in the long-term (which, by its nature, is much more difficult to assess)?

#### **4. Information on population-level consequences of climate change.**

Many studies indicate that climate change may lead or has already led to a shift in timing of breeding or migration. In that case, what are the consequences of a change in timing for a species and for species that interact with that species? When and how do the species that make up a community change? What are the consequences of such changes on species of concern? A change in prey species may be different than a change in competitor species or a change in a predator.

#### **5. Information on dispersal success and the impact of management actions.**

For example, what are the important characteristics of dispersal corridors? What are critical time periods for dispersal? What are the characteristics of movement itself (during which organisms may be at risk of mortality)? What determines success with respect to recruitment in new or unoccupied areas? That is, information on behavior (to disperse or not, where to disperse) is needed as well as success at recruiting into the breeding population.

This information will be needed in order to make decisions regarding translocation in addition to being an important component of viability. Population resilience will depend on successful dispersal.

#### **6. Essential characteristics of refugia and adjacent, transitional zones, both natural upland and human-influenced.**

A corridor or transition zone that serves an important ecological function for target organisms may also serve as corridor for a predator or an invasive/pest species. At the same time, transition zones can serve as conduits or barriers to dispersal (see #5, above).

#### **7. Information is needed on pollinators.**

More generally, what allows for connected subpopulations of plants in vernal pool and other priority habitat?

To address data gaps and sources of uncertainty noted above we recommend the following:

- **Encourage, support and update recommendations outlined in Chapter II with ongoing and innovative modeling and recalibration of climate change impacts.**

More specifically, the following is critically needed--the fruit of a next generation of landscape-wildlife-climate change models:

- i. Better predictions of future environmental conditions including extreme events.
- ii. Better information on tidal marsh restoration trajectory and how changing environmental conditions (as part of restoration) will affect wildlife and plants. Projecting change in marsh and tidal flats due to the climate, land-use decisions, and restoration need to be integrated.
- iii. Develop better predictive models of population growth and persistence in future landscapes, subject to unpredictable events. Such models would be spatially-explicit, incorporate demographic processes, incorporate climate change impacts, and could be used to consider management alternative actions. Ideally, such models would identify population bottlenecks, control points.

We close with a final recommendation, related to data gaps and uncertainty:

- **Establish a comprehensive monitoring program.**

A recommendation strongly urged by many workgroup participants is the establishment of a comprehensive monitoring program, in close coordination with other ongoing or planned efforts, including activities of the San Francisco Bay Joint Venture and the BAECC Climate Change Monitoring Project. In addition, a San Francisco Estuary Baylands monitoring program should be integrated with a Delta monitoring program, which is currently being developed.

Proposing details of such a program is beyond the scope of the BEHGU project so no specific actions are proposed here. Clearly, not everything can be monitored all the time, but we believe that the 32 case studies included here provide a starting point, and that their authors can begin to outline a regional, responsive monitoring framework. A comprehensive monitoring program is critical to guide management. The monitoring program would assess:

1. Changes in population viability and species distributions, as they reflect climate change and anthropogenic impacts.
2. Effectiveness in achieving the goals of a management action, both immediate changes in the environment as well as the more long-term (often species-specific) goals with respect to target organisms.
3. Specific wildlife and plant response to management action.

Such a feedback loop of data from the field and management actions is the foundation of many of the wildlife-focused recommendations detailed in Chapter 2 of the main report – *New Opportunities: How We Can Achieve Healthy, Resilient Baylands*. Species easily monitored and representative of other members of their community allow cost-effective and informative monitoring of trends in populations and better inference into underlying influences on those trends. Assessing status and response of target species (such as threatened or endangered species) may be an ultimate goal, but to assess environmental conditions, how they are impacted by climate change, and how the biological community responds, will require monitoring additional species that may be especially informative. The broad-based monitoring of bayland species must be integrated with monitoring the drivers of environmental change at appropriate spatial scales, as well as including information on habitat and landscape features, such as characteristics of the transition zones, that affect the viability of plant and animal species. Thus monitoring of habitat quantity, quality, configuration, and important features of the landscape, as well as the physical drivers, is to be integrated with monitoring of plants and animals, reflecting our knowledge of how changes in the environment are impacting wildlife.

Rare and sensitive species are difficult to monitor but are at elevated risk due the episodic nature of many expected climate change impacts. A responsive monitoring program that includes such species is needed, integrated with the broad-based monitoring of trends. The rarity of these species makes it hard to discern changes in their condition on the basis of broad-scale monitoring alone, but rapid, informed management actions may be essential for assuring their long-term survival.

## SUMMARY

Bayland plants and animals have evolved in the context of California's variable climate, across a landscape marked by dramatic changes in salinity and sea level. However, at present, suitable habitat has been severely reduced and remnant habitat is fragmented and degraded. Simultaneously, wildlife have been subject to an array of stressors such as contaminants, invasive species, human-associated predators (such as cats, crows, and gulls), loss of prey, unsuitable breeding conditions, and so on. Thus, the capacity of wildlife to adaptively respond to severe changes in habitat as well as to additional stressors, is already limited, and will be even more so in the future because: (i) future changes to habitat, due to climate change and land-use change, will occur at an accelerated pace, and (ii) extreme events, such as droughts, flooding, and storms, are expected to have increasingly severe impacts on wildlife.

Conservation of bayland plant and animal species in past decades focused on protection of individual populations in place, assuming stable vegetation structure and stable local environments. This assumption and conservation approach will be infeasible for the new sedimentary, submergence, storm, and salinity regimes of the 21<sup>st</sup> century estuary. To address these challenges we must focus on maintaining and promoting healthy, robust, interconnected populations, which are characterized by resilience in the face of dramatic changes in habitat and landscape configuration. Ensuring resilience means achieving high survival, reproductive success, and successful dispersal and recruitment of offspring into the adult population. For populations to be robust in the face of environmental change will require connectivity of habitat, on multiple time scales (daily, annual, decadal, etc.), and the ability of plants and animals to move effectively across the landscape, for example to colonize new habitat as it becomes available due to habitat restoration or climate change. Diversity within and among habitats is important, but it is especially important to sustain the dynamic processes that produce such diversity.

The long-term persistence of wildlife can be enhanced through management actions that reduce impacts of multiple stressors, thus increasing resilience to climate change impacts. Maximizing resilience entails enhancing recovery from catastrophic events (e.g., extreme storms) at the local scale as well as facilitating re-colonization of habitat where a species has previously been extirpated. Widely scattered, but connected, populations in appropriate habitats allow affected populations to recolonize or augment or "rescue" reduced populations. In species with limited mobility or with isolated habitats, translocations may be a necessary management action. Active and anticipatory management actions are needed to allow wildlife to weather the landscape-level changes and intensified stresses that climate change will bring to the San Francisco Estuary.

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**SUPPLEMENTARY MATERIALS**

**Table S1. Summary of case study results.**

Species or Group	Status	Bayland habitat, areas	Climate Change impacts	Other stressors, constraints	Sensitivity, Bottlenecks; other notes
<b>Salt marsh harvest mouse</b>	Endangered; trend not known. Higher numbers in some brackish areas.	Mainly tidal; also muted marsh; diked wetlands in Suisun. Throughout SFE; two subspecies.	Habitat loss; habitat fragmentation. Inadequate refugia from flooding, predation. Nest-flooding.	Loss of emergent vegetation or upland transitional zone (or very narrow band) a problem; Invasive plants.	Need tall and dense vegetation and/or upland refugia. Dispersal a limitation.
<b>Shrews:</b> Suisun and Salt marsh wandering shrew	Rare. Exception is Suisun shrew at Rush Ranch.	Tidal and diked marsh. Associated with marsh-terrestrial ecotone.	Flooding leading to food shortage. Habitat loss.	High overall mortality. Contaminants.	Little population data. Species survival depends on protection and expansion of ecotone.
<b>River otter</b>	Increasing; no special status	Creeks and reservoirs throughout SFE	Loss of denning habitat; change in prey due to change in salinity	Pathogens	Piscivores. Quite mobile. Populations not well-studied.
<b>Harbor seal</b>	Stable or increasing	Tidal rock, mudflats; throughout SFE	Loss of habitat for resting and nursing pups critical; shifts in prey, predators or pathogens	Disturbance, contaminants, disease	Sensitive to shifts in prey and to disease (especially to pathogens not previously encountered like phocine distemper virus)
<b>Ridgway's rail</b>	Federally Endangered; short-term decrease, increasing since 1990's	Saline tidal marsh; Central and South SF Bay; San Pablo Bay	Habitat loss and alteration; Inundation. Leading to nest failure and high predation	Human-associated predators. Affected by control of invasive Spartina.	Vegetative cover important. Need "in-marsh" refugia.
<b>Song sparrow</b>	Decline; species of special concern	Tidal marsh (and muted tidal); throughout SFE	Habitat loss and alteration; inundation from SLR and storms; shorter breeding season due to temp, precipitation	Nest-predation high (multiple predators)	Need to address low nest survival
<b>Black rail</b>	State threatened; increasing	Mid- to high-tidal marsh. In SFE except in South SF Bay.	Marsh inundation due to SLR and storms, in winter and spring. Leading to lower survival, reprod success	Invasive plants; predation by non-native predators; human disturbance; habitat fragmentation	Low dispersal; little known about this. Winter habitat requirements not known.

<b>Northern harrier</b>	Trend not known. Common.	Multi-habitat generalist. Throughout SFE.	Loss of habitat due to SLR; inundation of nests	Nest predation (foxes); contaminants (including lead)	Representative raptor in bayland habitat. Rely on upland habitat (affected by changes to that).
<b>Shorebirds: American avocet and Western sandpiper (WESA)</b>	WESA: apparent decline (or just shift). Neither species has special status.	Avocet: managed ponds and tidal marsh; breeder. WESA: tidal flats, winter-resident.	Loss of habitat (mudflat for WESE); prey may change, shift.	WESA: low reproductive rate. Avocet: low chick survival. Nest predation high.	WESA: vulnerable due to complex migration, not flexible; loss of managed ponds with high prey biomass. Avocet: California gulls a problem.
<b>Terns: Forster's (FOTE) and Least (LETE)</b>	LETE: Federally Endangered; FOTE: may be declining	LETE: breed on beaches, islands. Only a few locations. FOTE breed in marshes.	Fluctuation in prey (LETE: anchovy important). Water level affects nest success.	Mercury severe problem, causes egg, chick mortality	Most FOTE breed in former salt ponds, on islands.
<b>Dabbling ducks: multiple species</b>	Mostly increase in Suisun and San Pablo; pintail decrease	Managed wetlands, managed ponds . Winter throughout SFE; some breeding, in Suisun.	Reproductive success declines with temperature; salinity	Disturbance a concern; conditions on staging areas important; winter and spring habitat can influence reproductive success	Loss of diked/ managed wetland a concern (e.g., due to restoration); salinity can affect vegetation and invertebrate prey
<b>Diving ducks: multiple species</b>	Some species declining. No special status spp.	Open bay, managed ponds; mainly winter, throughout SFE	Loss of habitat due to SLR; change in prey distribution	Energetic (prey) constrained; disturbance, contaminants. Effects on breeding grounds	Herring roe very important for scoters; foraging depth important
<b>California toad</b>	Western toad declining; Calif. subspecies not known, common	Breed in freshwater; need upland refugia nearby. Throughout Bay Area.	Will be impacted by increase in temperature, salinity; decrease	Pathogens, UV-B radiation, salinity	Dispersal can be issue
<b>California Red-legged frog</b>	Fed. threatened; range has contracted	Tidal (brackish) marsh; breed in freshwater ponds	Inundation, storms; Salinity, Loss of habitat	Egg and larval survival extremely low; predation	Very sensitive to salinity. Limited by dispersal (need fresh ponds near marsh).
<b>Pacific herring</b>	Recent increase, after recent collapse	Shallow aquatic, North bay	Salinity shift away from spawning substrate	Ocean conditions, harvest	SF Bay one of few spawning sites. Economically valuable
<b>Delta smelt</b>	Drastic decline	Limited, Suisun and delta	Spawning window, Delta outflow	Entrainment, food web, predation	Very limited distribution; No other population. Sensitive.
<b>Longfin smelt</b>	decline	Limited, Suisun and San Pablo	Spawning conditions	Entrainment, food web, predation	Population in Washington State and maybe North

					Coast
<b>Longjaw mudsucker</b>	decline	Pickleweed marsh	Habitat loss,	unknown	Common elsewhere; limited range makes good local indicator
		widespread	desiccation		
<b>Tidewater goby</b>	extirpated	Estuarine lagoons	Re-establish?	Invasive species	In some coastal lagoons, including nearby
		San Pablo			
<b>Grunion</b>	extirpated	Sandy beach	Re-establish?	Human disturbance	Re-established briefly 2000-08; El Nino may facilitate
		North bay			
<b>Chinook salmon and steelhead</b>	decline	Vegetated	Upstream thermal stress	Ocean conditions, upstream conditions	River and estuary most important when ocean less productive
		Edge,			
		widespread			
<b>Dungeness crab</b>	increasing	Shallow aquatic, marsh channels, eelgrass,	limited	Ocean conditions,	Economically valuable; mobile and widespread; good broadscale indicator species
		widespread			
<b>Marsh invertebrates:</b> many	Various.	All parts of tidal marsh (including channels); throughout SFE.	Loss of habitat due to SLR, particularly loss of high marsh. Change in temperature and vegetation could affect invertebrates.	Invasive pepperweed an issue.	Some insects can tolerate inundation. These invertebrates not well studied. Important as prey for vertebrates. Depend on plants, phenology.
<b>Vernal Pool</b>	Rare. Includes plants, crustaceans, other inverts; amphibians.	Freshwater, ephemeral pools.	SLR will lead to habitat loss; saltwater intrusion. Increased temperature and decrease in precipitation reduce inundation periods.	Little remaining habitat; Isolated ponds	Manage for variety of pool size and depth.
<b>Spartina, Native and Invasive</b>	Native: wide spread, Invasive: Recent; nearly removed at present	Invasive: Most problematic in Central and South SF Bay; potential problem in San Pablo	Higher salinity and more inundation favor the invasive/hybrid.	Hybrid is very invasive. Control can be difficult. Invasive control can have impact to Ridgway's Rails.	Native is foundational species. Invasive responsible for major, physical alteration of habitat, loss of diversity.
<b>Submerged Aquatic Vegetation</b>	Widespread but only locally abundant. Possibly increasing in Suisun Bay subtidal habitat	Low turbidity shallow subtidal aquatic habitat (shallow bay, slough), sheltered marsh ponds, lagoons	Low turbidity (reduced suspended sediment concentration) favors increase growth;	High (polyhaline-euhaline) salinity favors only 1 genus ( <i>Ruppia</i> ), restricts range of both native	Unrepaired levee failure of diked subsided baylands may increase habitat as sea level rises. High wave energy at shoreline (marsh

				and non-native pondweeds ( <i>Stuckenia</i> , <i>Potamogeton</i> spp.)	peat erosion) and SLR may increase habitat. Increase of wigeongrass and pondweed; rapid change in relative abundance with seasonal salinity patterns is expected
<b>Low Tidal Marsh Plants</b> (grasses, sedges, tules)	Widespread; Foundational species. Few tall emergent species tolerate deep tidal flooding and dominate low marsh	Cordgrass dominates polyhaline-euhaline low marsh (to approximate MSL); tules, bulrushes, cattails, sedge dominate oligohaline to mesohaline low marsh (to approximate MLW)	SLR will likely convert intertidal marsh plains to transitional low-middle marsh zone or low marsh. Higher salinity during growing season favors cordgrass over tule, bulrush, sedge	Higher wave energy due to deepening bays (SLR) may restrict low marsh to wave-sheltered sloughs. Higher wave energy and salinity may expand cordgrass marsh with altered structure and ecogeomorphic function (selection for hybrid <i>Spartina alterniflora</i> traits in backcrossed populations)	High seed production, dispersal ability, rapid potential colonization of suitable salinity & disturbance regimes
<b>High Tidal Marsh Plants: annual forbs and graminoids</b>	Many reduced to rarity, with few & mostly isolated, small populations; reduced geographic range within Estuary	Mostly in or near remnant prehistoric tidal marshes in high marsh & transition zone	SLR will likely increase submergence of existing habitat (convert to middle marsh); constrained potential for new habitat to support populations	Mostly limited dispersal ability (broadleaf annual forbs), adapted to local dispersal in favorable, specialized sub-habitats	Most spp. have low potential seed dispersal. Potential rapid colonization of restored high marsh and transition zone habitat (assisted migration) with suitable structure (gaps or reduced competition with dominant perennials)
<b>High Tidal Marsh Plants: shrubs and perennial forbs (including T/Zone)</b>	Many spp. still common in brackish marshes, esp. N Estuary; only gumplant common in salt marsh today	Mature high brackish to salt marsh and lower terrestrial transition zones with complex soil and hydrologic gradients, and mature tidal channel banks (natural levees, well-drained bank edges)	Loss of habitat (high marsh submergence, erosion), increased frequency of marsh hypersalinity events (droughts)	Low tolerance of soil waterlogging; low to moderate salinity tolerance in most spp. Limited seed source populations in modern salt marshes (SFB)	Many species, high diversity, in relatively narrow but complex marsh edges with gentle gradients. Often associated with to mature or prehistoric tidal marsh remnants; limited extent and distribution, rarely

					restored.
<b>High Tidal Marsh Plants: perennial graminoids (including T/Zone)</b>	Many spp. still common in brackish marsh; only saltgrass common in salt marsh today	Mature high brackish to salt marsh plains to transition zones with complex soil and hydrologic gradients	SLR-induced submergence, increased frequency of marsh hypersalinity events (droughts); competition with more tolerant pickleweed; slower to recolonize than pickleweed	Limited seed source populations in modern salt marshes (SFB)	Often associated with mature or prehistoric tidal marsh remnants with peaty or sandy high marsh zones near seed source populations.
<b>Terrestrial/Marsh Ecotone Graminoid Plants</b> (Grasses, sedges, rushes; Transition Zone)	Few remnant or regenerated populations of perennial, sod-forming creeping grass, rush, sedge species bordering northern Estuary	Clayey to sandy loams on seasonally wet gentle slopes, above highest tide lines to spring high tide line	Frequent levee maintenance limits colonization and establishment.	Seed limitation of founder populations due to isolated and relatively small remnant populations. Cattle trampling and overgrazing destroys rhizome networks and reduced populations at tidal marsh edges	Increases following reduction of cattle trampling frequency. Low long-distance dispersal ability, high short-distance dispersal ability (clonal spread). Populations can regenerate following cessation of cattle trampling/overgrazing
<b>Terrestrial/Marsh Ecotone Plants psammophytes: sand plants</b> (Transition Zone)	Mostly extirpated due to historic loss of sandy tidal marsh transition zones; one sand-specialized endangered species reintroduced locally	Marsh-fringing barrier beaches, San Pablo Bay, Central SF Bay (historic core habitat) S Bay; some persistent or regenerated habitats provide refuge	Continued loss of habitat and coarse sediment (urban shoreline armoring, wave erosion in response to SLR)	Increased wave energy and erosion potential with deeper bay (SLR), reduced coarse sediment supply (armoring)	Potential for rapid recolonization of restored habitat by some common species; artificial reintroduction needed for rare species. Iceplant, perennial pepperweed can rapidly invade habitat

# Science Foundation Chapter 5

## Appendix 5.1 – Case Study

### California Red-Legged Frog (*Rana draytonii*)

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#### DESCRIPTION OF THE SPECIES

This case study considers the California red-legged frog (*Rana draytonii*) that is endemic to California and northwestern Baja California, ranging from central California, south to the mountains of southern California and east to the Sierra Nevada up to an elevation of 1,500 m (5,200 ft) (USFWS 2002). Molecular studies by Shaffer et al. (2004) demonstrated that *Rana aurora* and *Rana draytonii* are separate species with a small range overlap in northern California along the coast in Mendocino County. The California red-legged frog (CRLF) was historically more wide-spread than the current distribution, occupying greater portions of the western slope of the Sierra foothills and the Central Valley, the north Coast Range foothills, and the Southern Transverse Range and Peninsular Ranges of California. Current populations, though, are now fragmented, and the species has been extirpated from 70% of the former range, though reasonable numbers of frogs still occupy the north-central coast, around San Francisco Bay, and the central coast of California. The species presently is associated with all of the counties surrounding San Francisco Bay estuary, occupying many of the freshwater marshes that fringe, and tributaries that flow, into the bay: in Suisun Bay (i.e., near Suisun Marsh); in San Pablo Bay (i.e., near Sears Point at junction of Highway 37 and Lakeville Road); in South San Francisco Bay (i.e., near SF International Airport); central San Francisco Bay; and Contra Costa and Alameda counties (i.e., East Bay Regional Park lands; USFWS 2002). Specific locations identified in the North Bay include Point Reyes National Seashore, Suisun Valley, Green Valley, northwest of Fairfield, and the triangle or tri-city/county open space roughly defined by Interstate Highways 80, 680 and 780 between Vallejo, Cordelia, and Benicia. From the tri-city/county open space area and the hills north of I-80 (identified as the Jameson Canyon-Lower Napa River core recovery area) and in the Stebbins Cold Canyon Preserve in the northwest corner of the county (LSA 2009). As a consequence of the range contraction and population decline, the species was listed as Federally Threatened under the Endangered Species Act in 1996 and Critical Habitat was designated in 2010. The California Department of Fish and Wildlife lists California red-legged frogs as a Species of Special Concern.

California red-legged frogs are a large native frog (females up to 138 mm, 5.4 in), considered one of the largest native frogs of the western US. They are distinguished from sympatric congeners by exhibiting pronounced dorsal-lateral folds. Coloration is highly variable and not a diagnostic characteristic of the species, but can range from nearly brick red dorsum with dark spots to a brownish background color; hind leg and lower abdomen coloration is variable as well. Frogs are sexually mature at 3-4 years and egg-laying occurs from late November to April. In cooler climates near the coast, larvae (tadpoles) are able to overwinter, with metamorphosis typically occurring from 3-7 months after hatching (Fellers et al. 2001). Little

is known of the juvenile life stage. Adults are reported to live 8 to 10 years (USFWS 2002). The adult diet consists mostly of terrestrial and aquatic invertebrates (Tennant 1985), but they will also eat Pacific chorus frogs (*Pseudacris regilla*), California mice (*Peromyscus californicus*) and even smaller life stages of their own species (P. Kleeman, USGS, per obs.). Larvae likely eat algae.

CRLF occur in very diverse habitats which may differ among life stages and across the range. Adults are resident in dense, shrubby or emergent riparian vegetation near ponds or streams. Breeding habitat is associated with distinct water depth and hydroperiod habitat characteristics such that the water must be “shallow enough (and the hydroperiod must be long enough) to support submergent and/or emergent vegetation as oviposition substrate during egg-laying, and as a refuge from predation” (Collins and Collins 2007). They also can use deep ponds as long as there are shallow margins for vegetation to grow in (P. Kleeman, USGS, pers. comm.).

Physical characteristics that limit their distribution and habitat use include water temperatures higher than 29°C and salinity levels greater than 7.0 ppt for adults (Jennings and Hayes 1990). CRLF are freshwater species and have only slight tolerances to salinity. Adults and subadults avoid brackish waters, however, they will use freshwater habitats that are adjacent to brackish waters and sometimes become inundated with salt water during exceptional high tides or storm surges, at which time, frogs will retreat to refugia in adjacent upland habitat (P. Kleeman, USGS, pers. comm.). At Point Reyes National Seashore, eggs surviving to hatching have not been observed in water with a salinity > 2.0 ppt (P. Kleeman, USGS, pers. comm.). Critical habitat is defined by USFWS to include essential aquatic habitat, associated uplands, and dispersal habitat that connects essential aquatic habitat (USFWS 2002).

Across their range adult frogs vary in their habitat use; some may remain resident for an entire annual cycle, where as others may disperse to multiple habitats depending upon the reproductive status. Generally, though, they are limited to within 0.5 km of water. The species is slightly more tolerant of higher salinities than most frog species, and adult frogs have been observed in emergent vegetation in brackish ponds and streams adjacent to coastal beaches in Drakes Bay (Fellers and Guscio 2002).

Breeding occurs in many different aquatic habitats such as streams, deep pools, backwater areas, ponds, marshes, sag ponds, dune ponds and lagoons (USFWS 2002). Eggs masses are usually deposited on emergent vegetation, such as bulrushes (*Scirpus* spp.), cattails (*Typha* spp.), roots or twigs, but in ponds, they have been observed unattached (LSA 2009). At Point Reyes National Seashore, made-made stock ponds are used by breeding frogs, if there is sufficient emergent vegetation for cover and egg masses. Habitat required for breeding is limited to deep (> 0.7 m; 2.3 ft) still or slow moving freshwater (GOGA Draft management plan 2011), and Fellers (2007) noted that most breeding sites are deeper than 25 cm (0.8 ft). However, ponds that harbor predators such as non-native fish, crayfish or bullfrogs may limit the presence of CRLF. Frogs avoid terrestrial mammal and avian predators such as raccoons (*Procyon lotor*), river otters (*Lutra canadensis*), and wading birds when associated with dense cover, including dense emergent vegetation and also upland coastal scrub habitat (Collins and Collins 2007). Mortality is highest for the larvae to metamorphs with an estimated < 1-2% survivorship. This lifestage also has the largest variation in survivorship, and consequently are likely an important cause of natural variation in the population growth rate of the species (Biek et al. 2002).

Radio telemetry studies have found that CRLF can move up to 2.8 km (Bulger et al, 2003, Fellers and Kleeman, 2007). CRLF move towards breeding sites (ponds, marshes, slow moving parts of some streams) during the heavy winter rains. Some CRLF remain at breeding sites year round if habitat is suitable, while others will leave after breeding and spend much of the year in non-breeding habitat (riparian corridors, very small streams, marshes that may be too shallow to breed in).

## **CRITERIA FOR SELECTION OF THE SPECIES**

CRLF have been well studied over the last century (Storer 1925, Stebbins 1985, Fellers 2005), and more recently, in the San Francisco Bay Area there are several studies (i.e., Bobzien et al. 2000, Fellers and Kleeman 2007, Tatarian 2008). Recent studies have focused on mitigation associated with wetland restoration work, and others on potential impacts of climate change on wetlands due to sea level rise (NPS 2007). Because they are a federally listed threatened species, land managers have developed strategies for preserving and enhancing habitat for frogs, and frogs often serve as an umbrella species for fresh water habitats. The San Francisco Bay Area Network of National Parks selected the CRLF as an indicator species of these habitats and Golden Gate National Recreation Area is monitoring the species' distribution in some areas around the SF Bay Estuary (Fong, National Park Service, pers. comm.).

The species are of high conservation concern and considered highly vulnerable to climate change due to (i) anticipated impacts on their habitat (fresh water habitats near the ocean that may become too saline) and (ii) direct effects of tidal flooding and sea level rise on reproduction, as elaborated below.

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## **OTHER RELEVANT INFORMATION**

CRLF populations function as a metapopulation, a collection of sub-populations that exchange genetic information through individual dispersal events (LSA 2009). Subpopulations in breeding ponds may blink in and out of existence, with extinction and colonization being a function of the distances between ponds or aquatic breeding habitat, and the probability of a pond being occupied may be positively correlated with the distance to the nearest occupied pond (Marsh et al. 1999). Consequently, dispersal habitat corridors and distances are important for the species' survival. Frogs are year round residents, and though, they may disperse to forage in other habitats during the non-breeding season, dispersal has not been found to be greater than 2.8 km (Bulger et al, 2003, Fellers and Kleeman, 2007). Most frogs disperse less than 1.2 km (1 mi), and the median distance along the Redwood Creek corridor in Muir Woods NM was 150 m (Fellers and Kleeman, 2007). Dispersal is often from breeding sites to riparian corridors, and frogs may travel direct routes, even crossing open pasture land and roads.

In the San Francisco Bay Area, some populations are fragmented and isolated. The causes for population declines are many, and may include historical over-harvest for food, predation by non-native species, habitat alteration, pathogens and pollutants. Populations continue to decline in many regions; though in some areas, populations are stable.

While bullfrogs have frequently been called a threat, or even a primary cause of the declines, there is almost no direct evidence that this is the case (Fellers and Guscio 2002), and it is at least as likely that non-native fish (e.g., striped bass, green sunfish, catfish, mosquitofish) play a significant role in the decline of native ranid frogs (Hayes and Jennings 1986).

Restoration that targets creating freshwater ponds with dense emergent vegetation upslope from brackish water has been successful in attracting breeding frogs; however, creating new ponds next to brackish habitat may become brackish themselves due to SLR and this would not benefit frogs in the long-term (P. Kleeman, USGS, pers. com). Restoration sites in the San Francisco Bay Area include Giacomini wetland in Tomales Bay, Big Lagoon in Muir Beach, and Mori Point ponds in San Mateo County (NPS 2007).

## **REVIEW OF CLIMATE CHANGE EFFECTS ON THE SPECIES**

Amphibians generally, and frogs specifically, are sensitive to changes in temperature, moisture and salinity. Any change in these physical states may alter habitat, and in turn, further affect frogs. Changes in weather patterns (including predicted increased intensity and frequency of El Niño events) may change breeding habitat and behavior. And too, the phenology of breeding may change with climate changes. The timing of amphibian breeding is correlated to temperature and moisture cues, and studies in other areas have shown a trend towards earlier breeding with changes in climate (LSA 2009).

Winter rain seasons that begin normally and then stop prematurely, put egg masses at risk because CRLF often put their egg masses in shallower parts of ponds (presumably to take advantage of warmer water temperatures that allow the egg mass to develop more quickly), and if the rains do not continue, many egg masses can become stranded above the receding water line and perish. In tributaries that flow into the bay, egg masses could be at risk to increased flash stream flows that could dislodge and destroy egg masses during brief, high intensity storms. Larvae in smaller, more shallow ponds will be at increased risk due to rising temperatures and perhaps less precipitation overall that could cause ponds to dry out before the larvae can metamorphose and leave the pond.

As noted for other species, the availability of freshwater marsh habitat adjacent to tidal marsh habitat, and specific characteristics of tidal marsh habitat, including salinity, is expected to change in the short term (2030), mid-term (2050) and long-term (2100). Although tolerant of salinity, tadpoles are sensitive to salinity levels exceeding 4.5 ppt and egg masses exceeding 2.0 ppt (see above).

Salinity levels may be altered with either SLR or changes in the amount and timing of freshwater runoff. Ambient temperatures are predicted to increase with climate change which would lead to desiccation, alteration and fragmentation of wetland breeding and foraging habitat. Loss of freshwater marsh, ponds and riparian corridor habitat will fragment and depress populations.

Climate change models predict warmer, drier conditions, but also predict more intense and more frequent El Niño events: such conditions are likely to decrease survival of egg masses and tadpoles and increase uncertainty in breeding from year to year. Warmer, drier conditions may shorten the breeding season, and eliminate breeding sites due to desiccation and vegetation loss. Exposure to predators would be more likely if frogs have to disperse across less hospitable habitat and preferred aquatic habitat decreases in size and is fragmented. More intense El Niño events may produce more moisture intermittently to the benefit of frogs, but also may result in elevated sea levels and large storm surges, which would inundate tidal areas into brackish and freshwater marshes, resulting in breeding habitat change and fragmentation. Habitat fragmentation may also result from alteration of dispersal habitat between foraging and essential aquatic habitat.

Whether current freshwater marshes adjacent to tidal marshes will be able to survive increasing regional SLR, depends on whether accretion (organic and non-organic) can keep up with or outpace increasing sea levels. As water levels increase, can tidal marshes migrate to areas that are currently supratidal? That will depend on the availability of such areas, as well as on the ability for tidal marsh plant species to migrate as well (i.e., establish themselves in new sites). [Sentences above apply to all tidal marsh habitat, so may be moved to a more appropriate section.] Additionally, the ability of plants to tolerate higher ambient temperatures and desiccation will influence the ability of frogs to seek cooler areas under cover.

Elevated water levels due to extreme tides or storms events may damage egg masses in adjacent freshwater habitat and limit freshwater habitat. Timing of extreme water levels is also important: the complete egg cycle requires about 21 days and metamorphosis occurs in a minimum of 3.5 months. One inundation event of a freshwater marsh habitat because of major storms and extreme tides could raise the salinity enough to destroy an entire year of CRLF breeding effort by killing the eggs or larvae.

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## **OTHER STRESSORS**

Several factors may have a synergistic effect on and may confound ‘normal’ stressors to frogs, including development (urbanization, intensive agriculture, and excessive livestock grazing), UV-B exposure, predation, pathogens and pollutants. Any one stressor alone may not have a negative effect but synergistically may become significant when combined with other stressors. For example, pathogens may have minimal effects on a healthy population, but in the presence sub-lethal concentrations of pesticides, they may cause declines in populations in (LSA 2009).

Other synergistic effects include elevated pollutant concentrations or pathogen exposure in aquatic habitat where water evaporates with elevated temperatures. Tadpoles exposed to high concentrations of nitrite and nitrate fed less, swam less vigorously, and developed malformations of the body (LSA 2009). Other researchers noted strong correlations between frog declines and being down-wind of agricultural areas where large amounts of agrochemicals were applied (Davidson 2004).

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## **LIFE CYCLE CONSIDERATIONS AND POPULATION DYNAMICS**

Tadpoles and egg masses are vulnerable life stage for the species because they are more sensitive to changes in habitat, water quantity and salinity. However, an ecological sensitivity analysis on red-legged frogs noted that post-metamorphic vital rates and highly variable vital rates both have a strong influence on population dynamics (Biek et al. 2002).

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## **FACTORS THAT MAY AFFECT SPECIES RESILIENCE**

Population resilience will be enhanced if reproductive success, especially breeding habitat and survival of larvae are increased (see below for specific suggestions). Since the variability in the survival rates of larvae is so high, reducing mortality due to predation, desiccation especially during extreme temperatures and tides/water levels, and salinity will enhance population growth rates and thereby resilience.

Maintaining connectivity of habitat will help maintain resilience for dispersal and retreat. Particularly effective may be the establishment of restored freshwater marshes upslope of extant tidal marsh, especially marshes that are likely to be “sources” rather than “sinks” and not likely to be affected by SLR.

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## **LIKELY CLIMATE CHANGE IMPACTS AND RISKS**

Freshwater marshes adjacent to tidal marshes will change between now and 2100, but the magnitude of change is uncertain. The total amount of freshwater and tidal marsh habitat likely will shrink and current tidal marshes will migrate inland into areas that are currently upland (above Extreme High Water levels). A

management approach would be to maximize production of young and their successful recruitment into the breeding population as adults. Thus, maintaining essential habitats and connectivity for CRLF is essential.

Current levels of breeding success need to be increased despite future pressures from climate change (mediated through flooding of habitat) and predators (specifically, non-native aquatic species). The range of the species has contracted significantly over the past 50 years leading to the species federal listing. Any further decline in the habitat will jeopardize the survival of the species.

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## **MANAGEMENT ACTIONS TO BE CONSIDERED**

Management should focus on two key demographic rates: survival rates of tadpoles and over-winter survival. A priority for management is to increase reproductive success by reducing loss of freshwater habitat and reducing predation rates. Increased survival rates could be accomplished by reducing predator access or exposure to predators. This will be a challenge because numerous species are documented to prey upon frogs, including a variety of mammals, birds, and snakes. Reducing predation may serve to offset egg mass failure due to flooding of salt water. Removal of non-native aquatic predators such as bullfrogs and mosquitofish might increase survivorship of egg masses and juveniles.

Habitat restoration may increase survival rates and reduce predation rates. Restoration could include providing or enhancing inland and upslope habitat where frogs may currently reside adjacent to brackish water habitat. Such inland refugia may reduce the effects of flooding and provide refugia for adult and juvenile CRLF from extreme water levels during winter storms. Adults are tolerant of higher levels of salinity but may require more transition habitat to freshwater marsh. Planting native vegetation around marshes including coyote bush and berries also will increase the potential cover for CRLF to avoid terrestrial and avian predators such as raccoons and wading birds.

However, it is important to note that freshwater habitat adjacent to salt water marshes are marginal for frogs since they act as population sinks in the years that saltwater intrudes and destroys egg masses. Creating CRLF habitat in places where expected sea level rise could intrude into the created habitat is not a long term solution. Instead, creating ponds in inland habitat with corridors from brackish water areas in the same drainage would be a better long-term solution.

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## **UNCERTAINTY AND ITS SOURCES**

The future of tidal and adjacent freshwater marsh habitat is uncertain with regard to its location, extent, and specific characteristics. This uncertainty is due to uncertainty regarding climate change projections and decisions regarding land-use (maintenance of levees, conversion of current diked baylands to tidal marsh, maintenance of managed ponds, etc.) The magnitude, frequency, and timing of extreme water levels is difficult to predict but will have severe consequences for frogs. The demographic response of CRLF to climate change is not well known, especially with regard to survival rates since so little is known about the survival rates of various life stages.

## IMPORTANT DATA GAPS/NEEDS

Potential impacts of climate change on survival of egg masses, tadpoles and adults are not known. Small changes in survival rates can have substantial consequences for population growth (or decline) and population resilience. An important gap to address is information on environmental influences on survival of egg masses, tadpoles and juveniles. Since one ecological sensitivity analysis on red-legged frogs showed that post-metamorphic vital rates and highly variable vital rates strongly influence population dynamics, more research in this area would likely yield important information for management (Biek et al. 2002). The ability of CRLF to colonize newly restored habitat, or re-occupy habitat, is well documented; however, barriers to dispersal need to be identified and better understood. An inventory of the distribution of frogs in ponds adjacent to tidal marshes in the San Francisco Bay Area could guide management to identify potential restoration sites. Upland habitat in the same drainages where frogs currently exist in ponds (adjacent to tidal marshes likely to be inundated with SLR) could provide refugia and breeding habitat for frogs in the long-term future.

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# Science Foundation Chapter 5

## Appendix 5.1 – Case Study

### Marsh Macroinvertebrates

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#### DESCRIPTION OF THE SPECIES

This case study includes representative macroinvertebrates that live in the marsh plain, its associated channels and pannes (ponds), and the marsh-upland transition zone. While less visible than animals such as birds, invertebrates play important roles in physical and biological processes (e.g., burrowing activity and channel bank erosion, and detritivores breaking down organic matter) and are important food resources for higher trophic animals. Common invertebrates in these habitats include plant-hopper (*Prokelisia marginata*), beach hopper (*Traskorchestia traskiana*), pygmy blue butterfly (*Brephidium exilis*), inchworm moth (*Perizoma custodiata*), western tanarthus beetle (*Tanarthus occidentalis*), salt marsh mosquitoes (*Aedes* spp.; Maffei 2000a, Maffei 2000b, Maffei 2000c), crabs (native *Hemigrapsus oregonensis* and introduced *Carcinus maenas*), copepods, snails (e.g. native California horn snail *Cerithidea californica* and introduced *Ilyanassa obsoleta*, *Myosotella myosotis*), polychaetes (e.g. *Capitella* spp., *Eteone californica*, *Neanthes brandti*), small clams (*Macoma petalum*/*M. balthica*), and corophiid amphipods (Cohen 2011, Race 1982, Robinson et al. 2011). Some common species were described in detail in the San Francisco Bay Goals Project *Species and Community Profiles* (Goals Project 2000).

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#### CRITERIA FOR SELECTION OF THE GUILD

This broad group of organisms serves a variety of ecological roles. Some species are food for songbirds, shorebirds, waterfowl, fish, or small vertebrates (Goals Project 2000, Dean 2005, Robinson et al. 2011). Others pollinate plants or scavenge dead vegetation. The introduced isopod *Sphaeroma quoianum* can increase erosion of marsh channel banks through its burrowing (Galley and Levin 1999). Some species distributions are restricted to narrow habitats. Others provide a link between the marsh and upland habitats by traveling between them (Maffei 2000d). Many are poorly studied; therefore, this case study focuses on species for which we could find sufficient information. Climate change may create negative impacts on some species and positive impacts for others, depending on their ecology and the physical changes experienced by marshes in coming decades.

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#### OTHER INFORMATION ABOUT THE SPECIES

Each invertebrate species tends to have a preference for particular environments within the marsh, such as the marsh plain, channels, or pannes. For example, at China Camp Marsh in the North Bay, corophiid

amphipods and the bivalve *Macoma petalum* were found mostly in large channels, while the non-native mussel *Geukensia demissa* inhabited the lower sections of channels at the edge of the marsh (Robinson et al. 2011). While the marsh contains many species, a few of them tend to make up the majority of the numbers and biomass, and in San Francisco Bay marshes many of the most abundant species were introduced from other parts of the world.

Many factors influence where a particular species will be found in a marsh, including physical structure, predation, competition, larval settlement and survival, and disturbance (Kneib 1984). Race (1982) studied competition between California horn snails and introduced *Ilyanassa obsoleta* snails in San Francisco Bay. She found that the marsh pannes (small ponds) provide a refuge for California horn snails when they are outcompeted by *Ilyanassa* on mudflats. Life cycles also influence invertebrate distribution. Many species are dormant during the winter. Species that have larval stages may use different parts of the marsh at each stage, for example with an aquatic larval stage and adults living on vegetation.

Invertebrates have developed a range of physiological and behavioral adaptations for life in the marsh. These include adaptations to survive fluctuations in salinity, temperature, and water levels on a daily to seasonal basis (Parker et al. 2012).

Tidal marshes are highly productive ecosystems and marsh-derived organic matter and nutrients are important food subsidies for aquatic and coastal ecosystems (Teal 1962; Odum 1980). Relatively few experimental studies have been conducted on tidal marsh food webs and trophic structure in San Francisco Bay, though see Howe and Simenstad (2007). Stable isotope analysis is a common tool that uses naturally occurring elements to trace food web sources and pathways. Stable isotope analyses showed that food webs for marsh invertebrates (such as filter feeding mussels) depended mostly on inputs from the marsh itself rather than on nutrients from the bay, even in newly restored marshes (Howe and Simenstad 2007, 2011), suggesting that inter-marsh connectivity can play an important role in subsidizing food webs in early marsh restorations. Juvenile fishes forage on insects that fall into the water column from tidal marsh plant canopies, as well as tidal marsh-derived materials that end up in aquatic food webs (Howe and Simenstad 2011; Herbold et al. 2014).

Grenier (2004) used stable isotope analyses within a tidal marsh in San Francisco Bay and conclude that marsh plain macroinvertebrates have fairly simple food chains extending from aphids to wolf spiders (Grenier 2004). The diets of song sparrow (*Melospiza melodia*), California black rails (*Laterallus jamaicensis coturniculus*), and California voles (*Microtus californicus sanpabloensis*) appeared similar, based mostly on the aerial pathway (insects) rather than the benthic (ground) food web pathway (Grenier 2004). This food chain is spatially separated and distinct from benthic invertebrates in the mudflats, which are generally supported by benthic microalgae (Neira et al. 2005).

Food web studies have confirmed the role of cordgrass (*Spartina spp.*) detritus in the diet of surface and subsurface detrital consumers (Levin et al. 2006). Leaf-hoppers, which feed on plant sap, also had a stable isotope composition similar to *Spartina* cordgrass, indicating that cordgrass is their primary food resource. Several recent studies examined the effects of invasive hybrid *Spartina* (a hybrid between San Francisco Bay native *Spartina foliosa* and U.S. east coast species *Spartina alterniflora*) on invertebrate food webs. Hybrid *Spartina* created numerous physical and biological changes to invaded marshes that reduced species richness and density and further changed macroinvertebrate community structure from surface microalgae feeders (i.e., crustaceans and bivalves) to one dominated by belowground plant detritus consumers (oligochaetes; Levin et al. 2006, Neira et al. 2007). The reduction in bivalves, amphipods, and polychaetes within invasive hybrid *Spartina* zones is also a concern because these invertebrates are important food for migratory

shorebirds (Levin et al. 2006, Neira et al. 2007). Furthermore, while hybrid *Spartina* added more plant detritus into the marshes where it invaded, invertebrates such as crabs, clams, and mussels did not ingest it, indicating that invasive hybrid *Spartina* did not create a new food source (Brusati and Grosholz 2009) for the key invertebrate prey resources for migratory birds.

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## REVIEW OF CLIMATE CHANGE EFFECTS ON THE SPECIES

These species already live with periodic natural stresses in the form of tidal cycles and changes in salinity over a day or season. Therefore, the scale of climate change impacts may depend on whether changes are outside the range of variation these species already experience and their ability to adapt. In addition, some stresses caused by climate change may be long-term, such as loss of high marsh, while others may be more periodic, such as changes in inundation periods or seasonal salinity. The impacts described below for different climate change scenarios are educated guesses extrapolated from the available information on macroinvertebrate tolerances to temperature, salinity, and other environmental factors. The description below focuses on the scenarios that are most likely to have an impact on macroinvertebrate communities.

All scenarios: The effect of changes in temperature extremes and temperature fluctuations may depend on how the current climate of San Francisco Bay marshes compares to the physiological tolerances of particular species (Deutsch 2008, Kingsolver et al. 2011). In general, increased temperatures in water, sediment, or ambient air could each affect invertebrates, as the habitats of these organisms differ at different life stages or different parts of the tidal cycle. Not all changes may be negative; warmer temperatures could benefit insects in temperate latitudes by increasing population growth rates (Deutsch 2008). However, if marsh plants that are hosts for insect larvae shift their bloom periods, this could create a disconnection between when eggs are laid and when resources are available for larvae, reducing survival rates (Durant et al. 2007).

Two of the larger macroinvertebrates in salt marshes are native shore crabs (*Hemigrapsus oregonensis*) and invasive European green crabs (*Carcinus maenas*). Both species can tolerate a wide range of temperatures. Shore crabs live in estuaries from Alaska to Baja California, putting San Francisco Bay in the middle of their geographic range and temperature conditions. Therefore, it is likely that they will be able to tolerate increasing temperature, although that assumes that the San Francisco Bay populations are not constrained by being genetically adapted to current conditions here. European green crabs have invaded many places around the world and can tolerate temperatures up to 91° F and reproduce in temperatures up to 79° F (Cohen 2011).

Scenarios 2 and 4 (high sea level rise scenarios): The greatest impact to marsh plain macroinvertebrates will likely be from the conversion of high marsh to low marsh or mudflats due to sea level rise and marsh drowning (if sediment accretion rates do not keep pace with accelerated sea level rise). In most areas around the bay, the area of high marsh and marsh-upland transition zones will decrease as they become compressed between rising sea level and area that lacks upland migration potential, while overall intertidal habitat will likely increase (Strahlberg et al. 2011). Species that depend upon the higher marsh will have less habitat while those that can utilize lower tidal elevations will likely be less affected. Longer inundation periods could be detrimental to some invertebrates but beneficial to others (Robinson et al. 2011), depending on specific life histories. At Toley Creek marsh in San Pablo Bay, tidal inundation did not affect the number of adult insect species or their vertical stratification on marsh vegetation, suggesting that some insects can remain attached to vegetation when inundated (Cameron 1976). Attaching during inundation may be less energetically costly than migrating out of the marsh and back again with the tidal cycle. Many

insects live as adults in the marsh for only a few weeks and so must partition their available energy among development, reproduction, and dispersal in that short time period (Cameron 1976). Reduction in high marsh could reduce host plants available to some insect larva, such as the inchworm moth which lays its eggs on alkali heath (*Frankenia salina*). Loss of their habitat could affect the larval life stage greater than the more mobile adult stage for flying insects. Cameron's study (1976) focused on periodic inundation from tides but could be useful for understanding the possible effects of longer inundation with sea level rise.

Changes in vegetation structure caused either by temperature increases or greater inundation could also affect invertebrate community composition or survival. In a southern California marsh, reduced shade altered the species community and changed the composition of microalgae that provide food to invertebrates, increasing the proportion of insect larvae while decreasing amphipods and oligochaetes (Whitcraft and Levin 2007).

Marsh macroinvertebrates will also need to adapt to changing salinity conditions. Cayan et al. (2008) provide climate projections in which a reduced snow pack and warmer conditions will result in some years with greater runoff (as the result of increased precipitation), and other years with significantly reduced outflow and in particular a shorter runoff season due to reduced late season snow melt. The resulting scenarios indicate increasingly variable salinity in the San Francisco Bay-Delta. Unfortunately, we have little specific information regarding the salinity tolerances of many marsh macroinvertebrates.

Scenario 5 (extreme storm event): Extreme storm events may cause a die-off of vegetation that cannot tolerate increased periods of inundation (Thorne et al. 2013), causing negative impacts on invertebrates living within the vegetation at the time of a storm. This will mostly affect species or life stages that are not very mobile. High storm discharge into tidal marsh creeks could wash away sections of creek banks and displace invertebrates burrowing within them. However, this seems more likely to be a short-term effect on particular marshes.

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## OTHER STRESSORS

The marsh plain may receive inputs of pollutants from adjacent upland areas. Invasive plants could invade the marsh plain, possibly reducing host plants for some insect species, but the impacts would depend on how specialized those insects are on particular plants. Increased frequencies of extreme winter storms could affect everything from salinity to marsh erosion to pollutants washed downstream from uplands.

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## ENTIRE LIFE CYCLE AND INFLUENCES FROM OUTSIDE THE ESTUARY

The effects of climate change may impact different life stages in different ways, with varying degrees of severity. The habitat used by macroinvertebrates, especially insects, often differs among egg, juvenile, and adult stages, as do their tolerance for high temperatures, temperature fluctuations, and other climate-related changes (Kingsolver et al. 2011). However, few data exist on these factors for insects in general and most studies have focused on agricultural pests and pollinators (DeLucia et al. 2012), so it is difficult to extrapolate information for California marshes. Some species may be able to adapt to changes through altered behavior or physiology. In laboratory studies simulating climate change, insects adapted to changed conditions by adjusting their rates of feeding or development (Kingsolver et al. 2009). Invertebrates may also show evolutionary changes to climate change, through selection for particular morphological or physiological traits. Based on global phenology observations of wild plants Wolkovich et al. (2012)

estimates that spring leafing and flowering may advance at the rate of 5-6 days per degree C. These changes in plant phenology may ultimately lead to temporal mismatch with plants and their pollinators (Wolkovich et al. 2012) and insect emergence times with vertebrate consumers (such as juvenile fish, tidal marsh birds, and their young; Dunn et al. 2011).

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## FACTORS THAT MAY AFFECT SPECIES RESILIENCE

Whether marsh macroinvertebrates can adapt to climate change may depend on how the current climate and other abiotic features of the marsh such as inundation rates and salinity compare to each species' tolerance to those stressors. Deutsch et al. (2008) predicted that temperate insect communities would overall see little to some positive effect from increased temperatures, based on the fact that many of these species are already adapted to seasonal fluctuations and not living at the limit of their environmental tolerance. Of course, individual species could still suffer even if overall insect or other invertebrate populations increase.

Invertebrates' survival, growth, and reproduction depend partly on temperature. Species have a range of temperature tolerance and a smaller range of optimal temperature where growth, reproduction, or survival is maximized. Marsh species also experience variation in water and air temperatures that depend on the tidal cycle, freshwater inputs from riverine systems, and by season. A species' resilience to climate change may depend on where current conditions fall within its range of tolerance or optimal conditions. An additional complication is that while a species may inhabit a large geographic range, such as with *Hemigrapsus oregonensis*, local populations may be adapted to local conditions and their resilience to climate change will depend on how well they can tolerate a new range of variation. Finally, the overall effect of climate change on the marsh macroinvertebrate community will also be influenced by interactions among species, such as relative effects on predators versus their prey (Freitas et al. 2007). We lack data on these questions for species in San Francisco Bay marshes so we cannot make specific predictions.

Invertebrates could use behavior to adapt to climate change. Some species might be able to go into diapause to avoid high temperatures. For some species, such as intertidal snails, increased inundation due to sea level rise could reduce stress from desiccation. Others such as aerial insects may need to move higher on vegetation to avoid extended periods underwater. These are educated guesses and also based on the assumption that a variety of micro habitats will remain in sufficient quantities and quality as the sea level rises.

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## LIKELY CLIMATE CHANGE IMPACTS AND RISKS

- Habitat loss due to increased sea levels and inability to migrate marsh landward.
- Possible reduced survival of species and changing abundances and distributions as salinities become more variable under future climate change projections.
- Possible reduced survival of species for which increased temperatures are outside their temperature tolerances but fewer effects and possibly even increased populations of species that are not at their upper limit of temperature tolerance.

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## MANAGEMENT ACTIONS TO BE CONSIDERED

As this group, marsh macroinvertebrates, covers a wide range of organisms, any management action could have positive impacts for some and negative impacts for others. Designing marsh restoration or sea level rise adaptation plans so that future marshes will retain mid- to high marsh elevations and transition zones will help preserve the species and ecosystem functions in the marsh plain. Maintaining or adding connections among marshes, or between marshes and the upland transition zones, would likely benefit many species. See the transition zone chapter for more specific recommendations.

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## UNCERTAINTY AND KNOWLEDGE GAPS

There is much uncertainty about the effects on tidal marsh macroinvertebrates because they have not been well-studied in California. The magnitude of climate change impacts depends in large part on each species' tolerance to changes in temperature or salinity outside the range of current conditions.

Little is known about tidal marsh macroinvertebrates in California and much less is known about the impacts of climate change on these communities. The effects of climate change on macroinvertebrates in general are uncertain, although laboratory studies have provided some examples of what might occur (Deutsch 2008, Kingsolver 2011). Some research has been done in the course of monitoring the progression of marsh restoration projects (see for example Talley and Levin 1999), food webs for fishes and aquatic systems (Howe and Simenstad 2011, Herbold et al. 2014), and changes in food web dynamics due to invasive *Spartina* (Levin et al. 2006, Brusati and Gorsholz 2009). Perhaps the greatest data gap involves impacts to marsh macroinvertebrate distributions, densities, and community structure due to climate change and other environmental stressors and the resulting food web impacts to their vertebrate consumers.

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# Science Foundation Chapter 5

## Appendix 5.1 – Case Study

### Northern Harrier (*Circus cyaneus*)

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#### DESCRIPTION OF THE SPECIES

This fairly common bird of prey is a medium sized, slender bodied raptor with long wings and tail. Highly sexually dimorphic, females are larger than males, brown above and heavily streaked below. The smaller male is cadet gray above and mostly white below with sparse cinnamon flecking on the breast. Young birds are similar to the adult female, a rich brown above and broadly washed russet below but with less streaking. All ages and both sexes have a conspicuous white saddle across the rump in all plumages, a reliable field mark. The pale-plumaged adult male harrier may be confused with the White-tailed Kite, as their habitat preferences overlap, but the kite is smaller and more slender, shorter-tailed, tends to hover, and has black chevrons on its “shoulders.”

Northern Harriers are food and habitat generalists, although open habitats are selected. They are usually seen coursing low and buoyantly over marshland or grassland, ready to quarter and capture anything that moves—especially small mammals and birds. The dihedral and rocking-flight is similar to that of Turkey Vulture. The former common name, “Marsh Hawk,” well-describes the habitat preference. They are the most owl-like of our raptors, with facial disks that aid their acute hearing. Primarily “vole specialists,” but harriers frequently take other small mammals, birds (meadowlarks, quail), and snakes.

Northern Harriers nest at favored places around the San Francisco Bay but are much more common and widespread in winter. The nest is placed on the ground, often in tidal, brackish, or freshwater marshes or sometimes in a swale or moist pasture densely vegetated with sedges or rushes, limiting intrusion by mammalian predators. In spring, the males perform a dramatic courtship flight display, wheeling in an undulating roller-coaster flight in the vicinity of the nest site. Once the pair bond is established and the female is attending the nest, the male delivers prey and the pair exchanges the offering in the air, displaying the aerobatic skills of this unique predator.

Harriers are highly migratory, and it seems that in the non-breeding season, most adult females vacate the Bay Area whereas adult males remain throughout winter.

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#### CRITERIA FOR SELECTION OF THE SPECIES

The Northern Harrier is perhaps the most representative raptor that commonly forages in bay wetlands. It is a California Bird Species of Special Concern, priority 3, because of historic population declines due to loss or modification of wetlands and native grasslands (Davis and Niemela 2008). Because tidal marshes are an important nesting habitat, land management practices that are likely to disturb nest sites during the

breeding season (March–August)—e.g. mosquito abatement, marsh restoration projects, intrusion by researchers, hunters, and recreationists—should consider impacts to nesting harriers. “Ground nesting harriers are highly vulnerable to trampling by livestock, haying, plowing, flooding, and fire associated with some agricultural operations and management activities” (Davis and Niemala 2008).

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## OTHER INFORMATION ABOUT THE SPECIES

This raptor has a northern Holarctic distribution with only two subspecies. The New World subspecies (*C.c. hudsonius*) is the larger of the two. The diet is broad with distinct seasonal and geographical variation, dependent of cyclical prey abundance (Smith *et al.* 2011).

Nest sites in wet substrate are significantly more successful than those on dry substrate because of reduced predation on the former (Simons and Smith 1985).

Harriers are highly vocal, especially around the nest site and during flight display, uttering a high-pitched and repetitive “kekking.” Aerial displays over and around nest site are elaborate, including food exchange, talon gripping, and roller-coaster flight. Harriers are more prone to polygyny than other raptors (Smith *et al.* 2011). In the non-breeding season, harriers may gather in communal roosts where prey densities are abundant, sometimes in the company of Short-eared Owls. Foraging behavior and diet overlap between the two raptors, but the owl hunts more in twilight and night conditions. Terrestrial mammals are likely the most important predators of eggs and nestlings.

Mammalian predators include coyotes, foxes, skunks, weasels (river otter?), raccoons, squirrels, as well as deer and livestock by trampling. Avian predators include crows, ravens, and owls. Reptilian predators may include snakes.

Although the population suffered historic declines and habitat loss continues statewide (Davis and Niemala 2008), currently the Bay Area population appears to be relatively stable.

Home ranges vary or are adjusted with prey abundance (Temeles 1987). Reported home range sizes range from 170-15,000 ha; median 260 ha (*in* Smith *et al.* 2011). A nesting season study in Suisun Marsh (1987-1992) found harrier nest densities of 3.3 to 24.8 nests/km<sup>2</sup> (mean 8.4); the highest value was noted in 1987, a year of high vole abundance (Loughman & McLandress 1994). In another study at Grizzley Island Wildlife Area in 1987, 74 nests were found with 12.5 nests/100 acres in managed fields as opposed to 8.2 nests/100 acres in unmanaged uplands (Larsen 1987).

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## REVIEW OF CLIMATE CHANGE EFFECTS ON THE SPECIES

The loss of tidal marsh habitat associated with sea level rise is expected to reduce the extent of available nesting habitat around the bayshore (Stralberg *et al.* 2011, Thorne 2012, Thorne *et al.* 2012). Increased frequency of tidal marsh inundation associated with increasing strength of storm surges (Thorne *et al.* 2012) may increase flooding of nests resulting in reduced reproductive success (Stralberg 2010).

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## OTHER STRESSORS

- Predation of nest sites and nestlings by non-native Red Foxes (*Vulpes vulpes*) has been identified as a threat, as well as depredation by feral cats in more urban areas (Davis and Niemela 2008).
- Contamination of the food chain has unknown impacts, although organochlorine based pesticides resulted in reproductive failure and population declines in the mid-20<sup>th</sup> C. (Smith *et al.* 2011).
- Because birds make up a significant portion of the harrier diet, lead toxicity from upland game birds is a potential stressor (Martin & Barrett 2001, Pain *et al.* 2009), particularly in the Suisun region with the proximity of hunting clubs.
- Collisions with wind turbines is a potential stressor (e.g. Montezuma Hills), although flight patterns of harriers may put them at lower risk than other raptors.
- Even marsh nesting harriers use upland fields and open spaces for foraging. Conversion of uplands from agricultural production to urbanization is likely to have reduce habitat viability.

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## ENTIRE LIFE CYCLE AND INFLUENCES FROM OUTSIDE THE ESTUARY

Because at least a portion of the local population is migratory, and because the winter range extends into the southern U.S. and Mexico, land-use practices outside the San Francisco Bay watershed may affect locally nesting harriers. Transportation of contaminants from more southerly agricultural regions is a possibility.

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## FACTORS THAT MAY AFFECT SPECIES RESILIENCE

Northern Harriers are food and habitat generalists, therefore inherently resilient to some degree of environmental change, however the breadth of this resilience is not well understood. Harriers tend to avoid areas with intense human use, preferring wide-open spaces, a tendency resulting from generally large foraging territories.

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## LIKELY CLIMATE CHANGE IMPACTS AND RISKS

If predicted climate change impacts to tidal marshlands and adjacent bottomlands are realized (Stahlberg *et al.* 2010, 2011, Thorne 2012), availability of viable nesting habitat and prey species (e.g. voles in tidal marshes) are likely to be reduced.

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## MANAGEMENT ACTIONS TO BE CONSIDERED

- Minimize human disturbance near nesting areas and restrict public access and land management practices as necessary. (No buffer zone information available; 100-meter suggested depending on activity level).
- Reduce livestock impacts to nesting success by limiting access to nesting areas, especially during the nesting season.

- Maintain a mosaic of large undisturbed habitats for nesting and high value foraging areas—abandoned fields, wet grasslands, etcetera.
- Practice rotational grassland management (Suisun Bay, north shore) leaving some sections fallow each year.
- Delay haying and plowing until after fledging (mid-July) where feasible.
- Avoid raising water levels unnaturally in seasonal wetlands to prevent flooding nests.
- Investigate effects of environmental contaminants on harriers and prey populations.

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## **UNCERTAINTY AND KNOWLEDGE GAPS**

The uncertainty inherent in climate change modeling, land-use practices, etc. introduces a level of uncertainty to any prediction about future population viability of harrier population.

### **Knowledge Gaps**

- Long-term studies on survival, reproductive success, dispersal.
- Comparison of reproductive success of harriers nesting in natural habitats (wetlands, grasslands) versus anthropogenic habitats (croplands, pastures, etc.)
- Effects of environmental contaminants on reproductive success.
- The effects of repeated exposure to agrochemical in wintering areas outside the United States needs to be investigated (Martin 1987).

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# Science Foundation Chapter 5

## Appendix 5.1 – Case Study

### Pacific cordgrass (*Spartina foliosa*)

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#### DESCRIPTION OF THE SPECIES

In the San Francisco Bay, the low salt marsh zone has historically been typified by a single plant species, Pacific cordgrass (*Spartina foliosa*), which also is a co-dominant species in the mid-marsh plain. In recent decades, San Francisco Bay tidal marsh has been invaded by an introduced relative from the eastern coast of the United States, Smooth cordgrass (*S. alterniflora*), and subsequently by even more invasive hybrids formed between the Pacific and Atlantic species (Dahler & Strong 1997, Ayres et al 1999).

This case study describes the ecological role played by native Pacific cordgrass in providing habitat structure for a variety of species in San Francisco Bay, and the substantial changes in habitat structure and benthic invertebrate community composition caused by invasive *Spartina alterniflora* × *foliosa* hybrids (Levin et al. 1998, Levin and Talley 2002, Neira et al. 2006, Janousek et al. 2007, Brusati and Grosholz 2009, Grosholz et al. 2009a, Grosholz et al. 2009b).

The genus *Spartina* (cordgrass) is widely dispersed throughout temperate salt marsh systems of the world. It has been well-studied globally, both because it is a highly valued, habitat-forming foundation species within salt marshes to which it is native (Pennings and Bertness 2001, Warren et al. 2002), and because of the negative ecosystem consequences associated with human-facilitated introductions of *Spartina* species to areas in which they do not naturally occur (Neira et al. 2006, Strong 2009).

#### Species Description: Native *Spartina*

*Spartina foliosa* is a clonal grass species found from Baja California to Bodega Bay, that grows in tidal salt marsh in the San Francisco Bay (Hinde 1954, Callaway and Josselyn 1992, Daehler and Strong 1995, 1997, Vasey 2010). In the low elevation salt marsh zone that extends from mean high water to mean sea level, *S. foliosa* is the dominant native plant species (Mahall and Park 1976b). This foundation species forms the structure of the low marsh zone forming uniform fringes on tidal creek banks and broad, uniform plains on the edge of

marshes adjacent to tidal mudflats (Hinde 1954, Atwater et al. 1979, Baye et al. 1999). Midway through its tidal range, *S. foliosa* can be found co-occurring with annual pickleweed (*Salicornia depressa*), the only other native emergent plant species to occur at this range (Baye et al. 1999). As elevations approach mean high water, *S. foliosa* intergrades onto the middle marsh plain and is mixed in with stands of *Sarcocornia pacifica* (perennial pickleweed). However, *S. foliosa* presence quickly disappears above mean high water as pickleweed becomes the dominant plant species (Mahall and Park 1976a, b).

*S. foliosa* has a minimal presence in the brackish portions of the estuary, despite the fact that cordgrass grows faster in fresher waters (Watson and Byrne 2009). It is more productive at lower salinities with an increased growth rate, height, and germination success (Phleger 1971). Indeed, seeds of this species germinate at the highest rates in nearly fresh water (Crispin 1976). The lack of presence in fresher parts of the estuary is likely due to the increased presence of better competitors not tolerant of higher salinities (Atwater et al. 1979).

*Spartina foliosa* was historically abundant in all intertidal marshes from North Bay (San Pablo Bay) to South Bay (Atwater et al. 1979). Diking and filling of the marshes since the 1800s eliminated 85-90% of the tidal marsh habitat (Goals Project 1999). Since the 1970s, the spread of the *S. alterniflora* × *foliosa* hybrid swarm has caused additional decline of native *S. foliosa* populations in the South and Central Bays, as the native is directly displaced or assimilated by hybridization (Ayres et al 1999, 2004, 2009; Sloop et al 2009, 2011). Native cordgrass was essentially extirpated from some large areas of the East Bay, including the marshes of the Eden Landing Ecological Reserve. Large areas of *S. foliosa* are still intact in the North Bay and throughout the Don Edwards National Wildlife Refuge (DENWR) in the far South Bay, where only small patches of invasive *Spartina* hybrids have been found and treated (Ayres et al 2004a). *Spartina foliosa* does not currently have special regulatory status, but it is considered a critical part of the ecological functions of the native tidal marsh ecosystem (Vasey 2009).

Low marsh provides essential habitat for a variety of fish and wildlife species, such as Chinook salmon, Dungeness crab, bat rays, ducks, and wading and shore birds (Goals Project 2000). *S. foliosa*, the sole vascular plant in the native low salt marsh zone, provides foraging cover and nesting substrate for the endangered California clapper rail, *Rallus longirostris obsoletus* (Albertson and Evens 2000). None of these wildlife species, including the California clapper rail, are restricted to *S. foliosa*. Improved wetland conditions and increased availability of wintering habitat is proven to increase winter survival and improve body condition and subsequent reproductive success for waterfowl (Sedinger et al., 2011 and references listed therein) and other species. Native habitat with abundant, high quality foraging, and protected roosting qualities is an important attribute for birds and other species. Unvegetated mudflats provide critical shorebird feeding areas (Stralberg et al 2004). Native cordgrass habitat has been shown to result in a greater diversity of benthic organisms compared to areas dominated by invasive *Spartina* (Grosholz et al. 2009a,b); which in turn improves diet resources for motile invertebrates, which in turn enhances resources for forage fish and multiple trophic feed levels in the bay.

Due to its ability to colonize at low elevations, *S. foliosa* is an important component of tidal marsh restoration. It quickly recruited passively at many newly breached tidal restoration projects in San Francisco Bay. In Carl's Marsh in Petaluma and in Muzzi Marsh in Corte Madera, recruitment of *S. foliosa* occurred within four years of tidal breaching (Faber 2004, Tuxen et al. 2008). As low marsh establishes, a site can begin to support a suite of species that rely on native salt marsh vegetation to provide food web services and habitat structure (Dawson and Foster 1982, Vasey 2010). Vasey (2010) notes that as a foundation species, *S. foliosa*

has “a profound effect on tidal wetland functions such as succession, productivity, and habitat structure”. Vasey further states that Pacific cordgrass functions to “facilitate the occupation of tidal wetlands by a myriad of microbial, algal, plant, invertebrate, fish, and bird species.”

### Species Description: Non-native *Spartina*

*Spartina alterniflora* (smooth cordgrass) is widely dispersed on the eastern portion of the United States, occupying a home range that extends from Maine to Texas. It was initially introduced to the San Francisco Bay by the U.S. Army Corps of Engineers in the 1970s as part of a restoration experiment aimed at stabilizing dredge spoils (Williams and Faber 2001). By the 1980s, *S. alterniflora* had expanded beyond the original planting site, and sometime after began to hybridize with the native *S. foliosa* (Callaway and Josselyn 1992, Ayres et al. 2003, Ayres et al. 2004). First generation hybrids soon backcrossed with one or both parents, and with other hybrids, creating a broad range of hybrids, sometimes termed a “hybrid swarm.” Members of the hybrid swarm exhibit a range of phenotypes, with some individuals closely resembling one or the other of the parental species, and many others expressing traits that allow them to exceed either of the two parental species in their ability to colonize new areas and spread rapidly. It is the hybrids with these emergent properties that have become the most highly invasive.

Some well-documented emergent traits of the most invasive members of the *Spartina* hybrid swarm include:

1. Higher vegetative growth rates than native *Spartina*, achieving both greater heights (Ayres et al. 2003) and faster rates of lateral vegetative growth and clonal expansion (Ayres et al. 2003, Ayres et al. 2004). Several studies have documented hybrid *Spartina* to have 4-times greater above and below ground biomass than native *Spartina* (Neira et al. 2006, Brusati and Grosholz 2009).
2. Capability of growing over a greater tidal range than either of its parental species, thriving both above and below mean high water (Callaway and Josselyn 1992, Ayres et al. 2003), making these hybrid phenotypes able to invade and expand in tidal flats.
3. Higher fecundity than its parental species in terms of pollen production (Anttila et al. 1998) and viable seed production (Ayres et al. 2008b). Unlike either parental species, many highly fecund hybrids are also self-fertile (Sloop et al. 2009).
4. Tolerance to higher levels of variability in environmental conditions such as higher salinity, higher sulfide levels, and higher water resident time than its native counterpart (Callaway and Josselyn 1992, Ayres et al. 2003).

High rates of lateral and reproductive spread allowed for accelerating rates of expansion of these hybrid *Spartina* throughout the San Francisco Bay, as did their ability to pollen swamp stands of native cordgrass to create hybrid seeds. By the time full-scale control efforts by the Coastal Conservancy’s Invasive *Spartina* Project began in 2005 (Ayres et al. 2008a, Hogle 2008), hybrid *Spartina* had invaded over 800 acres throughout the estuary with the central and southern portions of the bay experiencing the densest infestations (Hogle and Olofson Environmental Inc. 2011). Newly breached restoration marshes and tidal flats, which are initially devoid of biotic resistance and competition from established native marsh plants, were among the areas most impacted by these invasive phenotypes. In tidal flats, hybrids resistant to tidal inundation and other pressures established self-fertile clones that then spread clonally and through seeds (Sloop et al 2011). . In restoration areas in which hybrids were allowed to establish, valuable shorebird

habitat was converted to a less diverse, hybrid *Spartina*-dominated marsh, similar to the *S. alterniflora* invasion of Willapa Bay, WA (Davis 2005). Hybrid *Spartina* formed dense monocultures with an absence of the channel complexity and the diverse zonation of mid-marsh that would likely develop in the absence of hybrid *Spartina* (**Figure 1**; Impact of Hybrid *Spartina* on restoration marshes: 47 projects totaling 1,600 hectares; ISP 2007a, 2007b, Ayres and Strong 2004a, 2004b, Ayres et al 2004b, Sloop et al. 2009, 2011).

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## CRITERIA FOR SELECTION OF THE SPECIES

The genus *Spartina* has been chosen for this case study due to the importance of native Pacific cordgrass, *Spartina foliosa*, in providing habitat structure in the low intertidal for a variety of species in San Francisco Bay, and the substantial habitat changes that the invasive *Spartina alterniflora* × *foliosa* hybrids have caused. *S. foliosa* is the foundational native plant species in the low salt marsh zone of San Francisco Estuary, providing critical habitat structure for many native and migratory fish, birds, and other wildlife. It is directly and immediately at risk from sea level rise at the same time that it is imminently threatened by extinction due to displacement by, and hybridization with, a non-native relative, *S. alterniflora*. State and Federal agencies, and other restoration advocates in the San Francisco Estuary, have invested approximately 24 million dollars to date and much effort to protect *S. foliosa* and the native marsh structure by eradicating the introduced *S. alterniflora* and controlling the highly invasive hybrids (*S. alterniflora* × *foliosa*). Adding complexity to this situation is that populations of the California clapper rail (*Rallus longirostris obsoletus*), an endangered marsh bird, have increased in the taller, denser patches of hybrid *Spartina*, leading some to speculate that the loss of *S. foliosa* and major changes to the tidal marsh structure caused by the spread of the hybrids might be justified, if it leads to recovery of this single endangered species. Finally, in recent years, as the expected rate of sea level rise has been adjusted upward, it has been suggested that the desire to preserve native *S. foliosa* and or restore a specific vision of tidal marsh should be abandoned, with the faster growing and more adaptive hybrid *Spartina* being allowed to dominate the marsh, accrete sediment, and raise marsh surface elevations at a rate potentially more in step with the rate of sea level rise.

Conscientious planning and management will be critical in the coming decades if we are to maintain and expand a healthy tidal marsh ecosystem in the face of sea-level rise. Without careful forethought, *S. foliosa* could be at risk of extinction due to loss of low-salt marsh habitat and genetic assimilation by the invasive hybrids.

Several studies characterize some of the impacts (physical effects, food web effects, etc.) in San Francisco Bay (Neira et al. 2005, 2006 Ecol Applications, 2007 Biological Invasions; Levin et al. 2006). Salt marshes on the Pacific coast generally consist of wide swathes of succulent pickleweed fringed with native Pacific cordgrass (*Spartina foliosa*). This is in stark contrast to Atlantic and Gulf Coast salt marshes which are typified by wide plains of smooth cordgrass, *Spartina alterniflora* (Macdonald and Barbour 1974). The introduction of *S. alterniflora* to San Francisco Bay and subsequent formation of hybrid *Spartina* with the native *S. foliosa* directly threatens the survival of the native Pacific cordgrass (Ayres et al. 2009), and has resulted in marshes that have been altered to more closely resemble eastern U.S. marshes, with both mudflat and pickleweed habitat supplanted by the hybrid *Spartina* invaders (Callaway and Josselyn 1992, Ayres et al. 2003).

The well-documented ecological consequences of the rapid expansion of hybrid *Spartina* is may be exacerbated by climate change. On the lower end of its tidal elevation range, there is a serious threat of conversion of mudflat habitat to hybrid meadows. Loss of mudflat equates to loss of foraging habitat for >500,000 migratory shorebirds that annually flock to the mudflats of the bay (Josselyn et al. 1990, Bildstein et al. 1991, Stralberg et al. 2004). Stralberg (2004) predicted between a 27 and 80 percent reduction in

mudflat area associated with uncontrolled hybrid *Spartina* populations. However, as we continue to reduce the acreage occupied by hybrids and control the most invasive phenotypes, it is not clear how hybrids that more closely resemble *S. foliosa* will respond to climate change and related impacts.

On the higher end of its elevational range, hybrid *Spartina* reduces marsh diversity by displacing perennial pickleweed, gumplant (*Grindelia stricta*), and saltgrass (*Distichlis spicata*) (Baye 2004, Point Blue 2013). The altered structure of the high marsh affects the distribution of resident songbirds (Nordby et al. 2009) and causes loss of habitat for an endangered species reliant on pickleweed, the salt marsh harvest mouse (*Reithrodontomys raviventris*) (Shellhammer et al. 1982). Avoidance of such hybrid *Spartina*-caused ecosystem stresses would likely benefit native species.

Hybrid *Spartina* also changes physical processes and food web structure in invaded marshes. Aboveground structure of hybrid *Spartina* slows water flow, trapping sediment and causing rapid rates of accretion not typical to native marshes (Neira et al. 2006). Plentiful detritus from hybrid *Spartina* is more abundant and decays more slowly than native detritus (Brusati and Grosholz 2009). Slower decay rates result in altered soil chemistry since native *Spartina* marshes have higher sediment respiration rates, shorter periods of ammonium resident time, and faster carbon turn over than their non-native counterparts (Grosholz et al. 2009a). These altered environments of hybrid marshes have been noted to change benthic invertebrate communities in terms of biomass, diversity, and functional group identity (Neira et al. 2006, Neira et al. 2007, Brusati and Grosholz 2009). The shift is most marked in converted tidal mudflat where the infaunal invertebrate community shifts from surface feeders that primarily consume microalgae to being dominated by belowground feeders that primarily consume plant detritus (Levin et al. 2006).

Because of concerns about multiple negative ecosystem effects of *Spartina* hybrids, the California State Coastal Conservancy and the US Fish and Wildlife Service's Don Edwards San Francisco Bay National Wildlife Refuge prioritized eradication of invasive cordgrass from the San Francisco Estuary through the formation of the Invasive *Spartina* Project (ISP; Strong 2009). In 2005, over 800 net acres of hybrid *Spartina* covered the estuary, but persistent control efforts by a region-wide coalition of ISP partners have reduced the hybrid presence by an order of magnitude (Hogle and Olofson Environmental Incorporated 2011), down to 39 net acres as of 2012 treatment. "Net acres" refers to the total footprint of invasive *Spartina* that is remaining bay-wide across the 25,000 acres of tidal marsh and 20,000 acres of mudflats that are monitored each year.

The endangered California clapper rail is an obligate tidal marsh bird found primarily within the San Francisco Bay Estuary. Clapper rail numbers increased from 1,040–1,264 in 1992-1998 (Albertson and Evens, 2000) to 1,425 ( $\pm 22$ ) in 2005-2008 (Liu *et al.*, 2009). The increase in clapper rail numbers occurred at a time when a hybridized, invasive *Spartina* spread rapidly throughout the Bay. Hybrid *Spartina* likely affected clapper rail numbers positively for two reasons: tall, dense hybrid *Spartina* provided increased cover for clapper rails reducing exposure to predators and tides; and hybrid *Spartina* converted mudflat to marsh habitat, allowing clapper rail populations to grow and expand into new areas. The subsequent reduction in hybrid *Spartina* has been accompanied by declines in California clapper rail populations, particularly in the Central and Southern San Francisco Bay. Review of the relation between hybrid *Spartina* cover and California clapper rail numbers suggests that the two are highly correlated, particularly at sites that lack natural features known to support clapper rails, such as channel networks and native *Spartina foliosa*. In the wake of the successful control of over 96% of the hybrid *Spartina* in the Bay, clapper rail numbers have since stabilized at most sites in the Bay (McBroom 2013).

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## REVIEW OF CLIMATE CHANGE EFFECTS ON THE SPECIES

Climate change predictions suggest that present-day marshes in the San Francisco Estuary will experience increased inundation times as a result of sea level rise (Parker et al. 2011). At the same time, the Estuary's salinity will become increasingly variable due to greater extremes in annual rainfall, decreased summer snow pack, and water divergence (Knowles and Cayan 2002). Salinity fluctuations causing alteration of plant communities is a well-documented phenomenon in historic records of San Francisco Bay. Soil cores and seed banks show an increase in native cordgrass presence during drier, warmer time periods over the past 5000 years (Goman et al. 2008). Additionally, an increase in the *S. foliosa* footprint was noted over the short time frame of 30 years during a survey of historic transects (Watson and Byrne 2012) in the estuary (prior to invasion). Currently, *S. foliosa* has a minimal presence in the brackish portions of the estuary despite the fact that it grows faster in fresher waters (Watson and Byrne 2009). It is more productive at lower salinities with an increased growth rate, height, and germination success (Phleger 1971). Indeed, seeds of this species germinate at the highest rates in nearly fresh water (Crispin 1976). The lack of presence in fresher parts of the estuary is likely due to the increased presence of better competitors and not tolerance of higher salinity (Atwater et al. 1979). Longer inundation times and greater water level fluctuations will likely increase and/or shift the footprint of *S. foliosa* and any remnant uncontrolled hybrid *Spartina* into the upper portions of the San Francisco Estuary..

While it is generally well understood how *S. foliosa* will likely respond to increased inundation and salinity, it is less understood how they may respond to increased carbon dioxide and increased temperatures that may result from climate change. Experimental studies have indicated that increased carbon dioxide in the atmosphere seems to favor plants with C4 metabolism, as does increased temperature (Rasse et al. 2005). Most wetland plants have this type of metabolism, including *Spartina*.

On the site level, differences in inundation, salinity, and plant interactions may impact patterns of plant zonation. Thus on a marsh to marsh basis, *S. foliosa* may decrease in abundance, and hybrids may or may not deviate from this pattern, depending on whether their intermediate characteristics confer an advantage over the native *Spartina*, especially in light of these added climate change-related stressors. It is not well understood what will happen to *S. foliosa* and *foliosa*-like hybrids within the confines of existing salt marshes. Mahall and Park (1976) suggest that Pacific cordgrass is excluded from the high marsh by increasing salinity while *Sacrocornia pacifica* is excluded from the low marsh by tidal flooding effects on seedling survival. However, if inundation increases, it would be expected that *S. foliosa* would increase in ecological footprint (Watson and Byrne 2009). It is not known how hybrids closer to *S. foliosa* on the morphological spectrum will respond, compared to either *S. foliosa* or more aggressively invasive hybrids.

The hybrid swarm initially contained four to six times greater genetic diversity than *S. foliosa*. As the invasion and subsequent treatment have progressed, there is some evidence that genetic diversity among the hybrids has declined, but average heterozygosity in the hybrids is still two to three times higher than in *S. foliosa* (Sloop et al 2011). This suggests that the hybrid swarm may still have an advantage over native cordgrass when it comes to adaptive potential in response to changing environmental conditions.

As sea level rises in reaches of the estuary with sediment supply deficits, it is likely that higher marsh zones will submerge to lower marsh zones (Warren and Neiring 2002). However, there are portions of the San

Francisco Bay that, because of the invasion and subsequent eradication of hybrid *Spartina*, have little to no native low marsh zones. As climate changes the San Francisco Bay and sea level rises, it will also be important to monitor *S. foliosa* in areas of the bay that are unaffected by hybrid *Spartina*.

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## OTHER STRESSORS

In addition to future projected climate change impacts, native *S. foliosa* is already under extreme pressure from existing stressors, including displacement by hybrid *Spartina* invasion. Additional stressors include habitat loss and fragmentation from development activities, urban sources of runoff and pollution, and nutrients.

Anthropogenic effects such as nutrient loading may variably affect *S. foliosa* as compared to hybrids. Tyler et al. (2007) found little response of *S. foliosa* to nutrient addition, while finding a significant response from hybrid *Spartina* at some sites and not others. Above-ground biomass did not respond to nitrogen addition along the marsh edge at Robert's Landing, and at Elsie Roemer Marsh there was no response to nitrogen in the meadow. There was also very little response in the below-ground biomass of hybrids. Ryan and Boyer (2012) found that enhanced nitrogen levels increased the height and cover of *Sarcocornia pacifica* and will likely further increase its dominance, as will increased salinity, to the detriment of co-occurring marsh plain species (e.g. *Distichlis* and *Jaumea*) (Ryan and Boyer 2012).

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## FACTORS THAT MAY AFFECT SPECIES RESILIENCE

In the absence of hybrid *Spartina* and other anthropogenic stressors, native cordgrass is a resilient species capable of growing at higher salinities and lower inundations than the majority of marshland species of San Francisco Bay. However, invasion pressure and pollen transfer from remaining hybrid *Spartina* still threatens this resiliency. Thus, controlling hybrid *Spartina* is key to maintaining native marsh structure. Genetic introgression, habitat degradation, and sea level rise are the biggest factors affecting native cordgrass populations in the Bay. Resilience can be substantially increased by continuing to invest in protection of existing marshes, continuing the eradication of hybrid *Spartina*, and restoration of new sites in order to increase the footprint of native cordgrass in the bay.

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## LIKELY CLIMATE CHANGE IMPACTS AND RISKS

Likely climate change impacts include drowning of native cordgrass due to sea level rise. This would occur at a bay-wide scale and has a high likelihood of occurring if there aren't actions to allow the migration of the marsh to higher elevations that meet native cordgrass habitat requirements.

As the Bay salinity becomes more variable, there will likely be a shift in geographical distribution, and *Spartina foliosa* may displace brackish plant species in the fresher parts of the estuary, such as Suisun Bay (Byrne et al 2001).

Precipitation and temperature extremes and/or changes will affect the timing of annual emergence and seed set success for plants, including *Spartina foliosa* and remaining hybrid genotypes.

Changes in climate conditions, hydrology and phenology will likely cause major shifts in tidal marsh community composition, potentially creating new competitive challenges for the native *Spartina* with new arrivals of low elevation tidal wetland species or *Spartina* hybrids (Ayres et al. 2008).

## MANAGEMENT ACTIONS TO BE CONSIDERED

- Develop a long-term funding strategy to complete elimination of non-native and hybrid *Spartina* from the San Francisco Estuary, thereby restoring native *Spartina foliosa* to its dominant position in the lower marsh zone.
- Develop an efficient and effective annual, long-term monitoring and rapid-response program to assure protection from re-emergence of the invasive.
- Strategically assist establishment of native low marsh (structure and vegetation) on the leading tidal edge of the bay during sea level rise. This may include assuring establishment of non-vegetated areas of low gradient where *Spartina foliosa* can readily establish as the Bay expands.
- Establish and maintain a diverse array of *Spartina foliosa* stock and managed tidal marsh reserves to hedge against the possibility that sea level rise could occur too rapidly to allow survival of natural *Spartina* marshes.
- Continue aggressive revegetation of tidal marsh restoration projects with *Spartina foliosa* to assure establishment of an ample and robust source of native propagules.
- In areas in which native cordgrass has been extirpated, implementing revegetation efforts with native cordgrass is a critical management action to enhance marsh biodiversity and native structure. Native cordgrass provides high-value native habitat at the lower tidal elevations.
- Work with current living shoreline projects that seek to buffer shorelines from wave attenuation, sediment stabilization and accretion. Determine how native *Spartina* can be incorporated into these designs.
- Conduct research to gain a better understanding of how *S. foliosa* and extant hybrids will interact with climate change effects in both the north and south bay.

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## UNCERTAINTY AND KNOWLEDGE GAPS

There are no existing data that document the ability of invasive *Spartina* to provide *long-term* habitat benefits for California Clapper Rails, or the long-term ability of *Spartina* to keep pace with a projected six to eight feet of sea level rise over the next 100 years.

### Knowledge Gaps

- ability of hybrid *Spartina alterniflora* x *foliosa* and native *Spartina foliosa* to keep pace with sea level rise
- value of plant height vs. stem density
- value of much reduced root and rhizome density which does not exclude infaunal invertebrates like hybrid roots and rhizomes
- deposition rates in *S. foliosa* marshes vs. extant hybrid *Spartina*
- unanticipated impacts from planting/not treating invasive hybrid *Spartina*
- issues with comparing east coast systems/*alterniflora* to west coast systems
- long-term negative impact of hybrid on clapper rails

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## Science Foundation Chapter 5

### Appendix 5.1 – Case Study

# California Ridgway's Rail (*Rallus obsoletus obsoletus*)

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#### DESCRIPTION OF THE SPECIES

The clapper rail (*Rallus longirostris*) recently was split into two sister groups (Chesser et al. 2014) on the basis of phylogenetic analyses (Maley and Brumfield 2013). The original grouping is now represented on the East Coast of North America by the Clapper rail (*Rallus crepitans*) and on the West Coast by the Ridgway's rail (*Rallus obsoletus*). The California Ridgway's rail (*Rallus obsoletus obsoletus*; hereafter California rail) is the largest of 3 Ridgway's rail subspecies which also include *R. o. levipes* (light-footed rail) and *R. o. yumanensis* (Yuma rail) (Eddleman and Conway 1994). The California rail is also the largest resident terrestrial vertebrate that lives solely in San Francisco Bay salt marshes. Formerly present in salt marsh along the California coast from Morro Bay to Humboldt Bay, by the early 1970s the California rail was breeding only in Elkhorn Slough and San Francisco Bay (Gill 1979). Population declines throughout the 19<sup>th</sup> and 20<sup>th</sup> centuries were attributed predominately to land conversion, development, and fragmentation of marshlands (USFWS 2013). However, exotic predators such as red fox (*Vulpes vulpes*) and Norway rat (*Rattus norvegicus*) are also implicated in population declines (USFWS 2013).



Tidal inundation of habitats used by California rails that reduce refuge availability and increase susceptibility of nests to flooding (Overton 2014).

California rails are facultative obligates of salt marsh, but do occur in brackish water marshes such as those in Suisun Bay and the mouth of Coyote Creek at lower apparent densities. Suitable vegetative cover, large marsh size, and extensive channelization appear to be the primary factors influencing presence and/or density of California rail populations (DeGroot 1927, Harvey 1988, Foerster et al. 1990, Albertson and Evens 2000). Mid-marsh elevations may also be more strongly associated with large California rail populations than lower or higher elevation marshland (Liu et al. 2012). They are omnivorous and not selective in their diet, but the majority of food items consumed appear to be mollusks and crustaceans (Moffitt 1941, Albertson and Evens 2000, C. Overton, USGS, unpublished data).

California rails are monogamous and highly territorial, particularly during the prolonged breeding season from March through August (Albertson 1995, Zembal et al. 1989, Zembal and Massey 1987, Applegarth 1938). Two types of nests are constructed from wrack and marsh vegetation. The first is built around the stems of cordgrass (*Spartina* spp.; native or invasive) and will often float up the stems with the tide; the second is often woven into branches of gumplant (*Grindela humilis*) and/or perennial pickleweed (*Sarcocornia virginica*) and remains in place even during high flood tides (Schwarzbach et al. 2006, Harvey 1988, Foerster et al. 1990). Both adults tend the nest and care for young, often building platforms for broods to rest on during high tides (Eddleman and Conway 1994). California rails may construct multiple dummy nests, often in close proximity to active nests (Gaines et al. 2003).

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## CRITERIA FOR SELECTION OF THE GUILD

Like many listed and special status species occupying tidal marshlands of Northern and Central California, threats to the endangered California rail are largely the result of anthropogenic changes to sensitive habitats that have a limited distribution (Goals Project 1999, Albertson and Evens 2000, USFWS 2013). California rails provide a unique case study for effects of climate change in the San Francisco Bay Estuary. Unlike other tidal marsh species in San Francisco Bay, the California rail is closely tied to tidal channels and sloughs for foraging and nesting and typically uses lower elevation habitats more frequently than other sensitive species.

Substantial effort toward understanding California rail population change and habitat requirements has been undertaken by a wide variety of interested parties (e.g., Point Blue Conservation Science, US Fish and Wildlife Service, California Department of Fish and Wildlife, East Bay Regional Parks District, State Coastal Conservancy, Invasive Spartina Project, and U.S. Geological Survey). Although populations have roughly stabilized since approximately 2005 (McBroom 2012, Liu et al. 2012), low abundance, limited distribution, and apparent restricted niche occupied by California rails suggest management options to mitigate effects of climate change may be limited. Nevertheless, the California rail remains an icon of San Francisco Bay salt marshes; widely sought after by naturalists and the only top consumer that is a year-round resident in the salt marsh food web. As such, California rails serve as indicators of the overall health of the San Francisco Bay ecosystem.

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## OTHER INFORMATION ABOUT THE SPECIES

California rail populations are fragmented but widespread throughout the intertidal salt marshes of San Francisco Bay. Population abundance recovered from a few hundreds of individuals in the early 1990s to nearly two thousand in the early 2000s (Albertson and Evans 2000, Liu et al. 2012). Between 2005 and 2011, coincident with habitat modification to remove invasive *Spartina* (*Spartina foliosa* × *alterniflora*),

California rail populations declined to just over one thousand (Liu et al. 2012). This decline occurred predominately in the South Bay where invasive *Spartina* infestation was greatest although North Bay populations declined to a lower extent (Liu et al. 2012). A large bay-wide decline in California rail detections in 2008, even in areas unaffected by invasive *Spartina* eradication, highlights the sensitivity of this species to annual variation in ecological conditions.

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## REVIEW OF CLIMATE CHANGE EFFECTS ON THE SPECIES

Our framework for assessing the impacts of climate change on California rails stem from two principal assertions: 1) The primary cause of declining populations of California rails in San Francisco Bay is low survival and reproductive success; and 2) vegetation cover which provides escape cover from predation and nesting substrate appears to be limited during the winter and periods of increased tidal inundation (Overton et al. 2014). The presence of invasive *Spartina*, which grew taller, had greater stem density, and senesced less than native *Spartina foliosa* (Callaway and Josselyn 1992) increased California rail survival rates which declined following herbicide application to survival comparable that in natively vegetated marshland, and were among the lowest recorded for any rail species (Overton et al. 2014). To the extent that these patterns hold true throughout the San Francisco Bay Estuary, any changes to tidal inundation patterns or the extent and structure of intertidal vegetation are likely to have the large impacts to California rail survival and reproduction. These impacts could result could result from global climate changes, regional climate patterns such as El Niño/Southern Oscillation (ENSO), or local weather patterns.

The effects of tide and vegetation structure on California rail survival is becoming clearer (Overton et al. 2014), but the nesting ecology of California rails is still poorly understood. Estimates of fecundity are low (1.9-2.5; Schwarzbach et al. 2006) yet consistent with contemporary population declines (Liu et al. 2012). Flooding appears to have been a minor risk in many marshes during years with moderate tide levels (Schwarzbach et al. 2006), but may increase during storms, following eradication of invasive *Spartina* and sea-level rise. The 28-30 day incubation period of California rails is longer than that of conspecifics (Eddleman and Conway 1998) and ensures that at least one cycle of “spring” high tides is experienced during incubation. The risk to California rail nests posed by tidal inundation is moderated by the choice of nesting site and the structural characteristics of vegetation where the nest is placed. Climate projections suggest that the San Francisco Bay region will experience reduced precipitation and changes to the timing of water flow through the Sacramento-San Joaquin Delta in addition to increased salinity in portions of the estuary (Chapters 2, 3, and 4). Both increased salinity and increased inundation time affect height of salt marsh vegetation (Woo and Takekawa 2012) and reduce the value of existing habitats as tidal refuge or as nesting substrate. Expansion of saltier waters could result in expansion of suitable California rail habitats through Suisun Marsh and upstream into current freshwater habitats within the Delta. Furthermore, increased variability in weather patterns and extremes resulting in more severe flooding and drought cycles are likely to occur. As such, there may not be sufficient resilience in California rail demographic rates to adjust to any increased pressure resulting from climate change. To the extent that marshes shift toward lower elevation and/or shorter vegetation structure, California rail survival, and likely recruitment, will be reduced. To the extent that suitable salt marsh vegetation expands up the Estuary, increase in the potential distribution of California clapper rails is possible. Both beneficial and detrimental impacts to California rail habitat are likely to occur under projected climate change scenarios and a management framework to compensate for these contraindicating effects is urgently needed.

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## OTHER STRESSORS

Several authors have indicated a positive link between California rail population density and marsh area (Garcia 1995, Liu et al. 2012). However, most marshes in San Francisco Bay are insufficiently large to provide the conditions which apparently promote increased California rail density. Although increase in California rail density is greater in small marshes, for a given increase in marsh area, than larger marshes, smaller marshes still have lower density than larger marshes (Liu et al. 2012). Since most marshes in San Francisco Bay are less than 50 ha, California rail populations in these areas are more at risk of local extinction due to lowered population size. The potential for inverse density dependence (i.e. Allee effect) to accelerate impacts due to climate change should be recognized. As population size drops, individual California rail survival probability may decrease due to increased exposure rate to predators (i.e. inverse density dependence). This could accelerate population change and drive small populations extinct. If marshland characteristics change and reduce the population of California rails, increased variability in population growth rates could result (Gilpin and Soule 1986). Although California rails can be highly mobile and redistribute to virtually any location within their range (Casazza et al. 2008), the frequency of such occurrences is exceedingly uncommon, suggesting that increased isolation and fragmentation of populations is likely to result as marshland is lost to sea-level rise and concomitant habitat change. Reduced spatial interdependency among populations is likely to affect both individual population viability within a single marsh and increase the consequences of catastrophes (e.g. oil spills) for the entire species (Hanski 1982).

However, an alternate (if interim) situation may also occur. Conversion of existing high and mid-marsh habitats to low marsh habitats as a result of sea-level rise may alter existing vegetation communities leading to vegetation structure of greater overall height as tidal inundation patterns become more favorable for native monocot vegetation, i.e. *Spartina foliosa*. While conversion of short mid-marsh vegetation to taller low-marsh vegetation may create increased nesting, refuge, and foraging opportunities for California rails in the near term, senescence of monocots during the winter may still result in seasonal bottlenecks to population dynamics creating an ecological trap that could exacerbate population declines. In addition, conversion of low-marsh vegetation to mudflat and homogenization of habitat types could reduce the ability for California rail populations to respond to random, large perturbations (i.e. catastrophes). Vegetation type conversion would place an increase risk during El Nino cycles, particularly those occurring in conjunction with storm events, when water levels can increase dramatically. Under such situations, likely outcomes include increased nest failure and reduced survival, particularly during the winter. In order to maintain resilience of California rail populations it is suggested that managers maintain heterogeneous marsh structure, and maintain or increase the number or size of marshes in San Francisco Bay (USFWS 2013).

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## ENTIRE LIFE CYCLE AND INFLUENCES FROM OUTSIDE THE ESTUARY

While California rails remain resident within San Francisco Bay throughout the year, this does not suggest that influences from outside the Bay do not affect California rail populations. In 2008, California rail populations declined dramatically throughout San Francisco Bay (Liu et al. 2012). In the same year, apparent declines were noticed in the light-footed Ridgway's rail surveys in Southern California as well (R. Zembal, *pers. comm.*). While a mechanism that would result in synchronous population dynamics between the two conspecifics has not been formally evaluated, very large scale climate or ocean patterns could be operating to influence both subspecies. Furthermore, changes in the predator community occur in the winter when vegetation and tides create the most extreme conditions for refuge cover used by rails. Climate-induced changes to the distribution and abundance of seasonal predators (e.g. short-eared owls, *Asio flammeus*) could propagate through the food web to impact California rails.

## **FACTORS THAT MAY AFFECT SPECIES RESILIENCE**

### How Can Resilience be Increased or Maintained

The structure and function of marshlands are a direct consequence of the tidal dynamics and sediment supply available to build and/or maintain marshes. Scenario models included within this update (Chapter 2 in this report) recognize that suspended sediment supply in San Francisco Bay may continue to decline and accelerate transition of marshland to lower elevation ecotypes. Even if marshes were able to maintain characteristics, elevated water levels would reduce the suitability of existing vegetation to provide refuge cover. The potential impact of short-term storm events on California rail should also be recognized. Scenarios that incorporated a 2045 storm event coinciding with winter king tides predicted sea levels that reached 75 cm higher at the Golden Gate Bridge than non-storm king tides occurring in 2000 due to combined effects of sea level rise and storm surge (Chapter 3, this report). Under these scenarios local inundation in other regions of the Bay may be more severe due to increased tidal range (e.g. South Bay), proximity to watershed runoff (e.g. Petaluma River), or local wind patterns that result in significant wind-driven waves (2 m or more). The impact to the California rail during these periods may be severe as intertidal refuge habitats are inundated and birds are forced to search for alternative cover in upland-transition ecotones or occupy suboptimal cover within the marsh with increased predation risks. If these changes progress, California rail survival and nest success will likely be reduced. Either demographic process could be critical for the small fragmented populations in the remaining marshlands surrounding San Francisco Bay which may already be near maximum reproductive capacity (Schwarzbach et al. 2006). It remains unclear whether California rails have a relatively r- or K- selected life history since both survival and fecundity appear low. As such, the undetermined sensitivity of populations to changes in either survival or recruitment rates confuses where the focus of conservation efforts should be placed (Martin and Wilson 2011). Further confounding appropriate management actions is uncertainty on the effectiveness of intervention strategies to improve demographic rates. Reduced reproductive success due to contaminant exposure for example may be a more intractable problem than insufficient nesting substrate or predation risks (Schwarzbach et al. 2006).

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## **LIKELY CLIMATE CHANGE IMPACTS AND RISKS**

### Magnitude and Likelihood of Effects, Bottlenecks, and Other Stressors

Climate change scenarios indicate that tide levels will increase and the amount and timing of precipitation occurring in the San Francisco Estuary will change. Both of these patterns will influence the distribution and characteristics of marshland vegetation and affect California rail populations. California rail survival and reproductive output are likely to be reduced as marsh area is lost due to drowning and/or insufficient sediment supply or as vegetation communities shift in response to changes in inundation frequency and duration (Bockelmann et al. 2002). Increase in the frequency or severity of droughts and storms may also influence vegetation characteristics resulting in lower quality habitats for California rails. The effects of these changes may be most pronounced during the winter when California rail populations already experience a bottleneck in survival rates (Overton et al. 2014). Reduction in marshes that increase fragmentation or isolation of remnant California rail populations may increase susceptibility of the entire species to catastrophes or reduce the spatial buffering that connected populations provide and lead to increased risk of extinction (Hanski 1982, Gilpin and Soule 1986).

## MANAGEMENT ACTIONS TO BE CONSIDERED

- Retain existing, actively restore, or promote establishment of habitat features that provide tidal refuge for California rails.

Projections of sea-level rise suggests that the specific locations that currently provide tidal refuge habitat are likely to change as upland margins become refuge habitat and existing refuge habitat becomes intertidal habitat. While it is possible that the projected climate impacts outlined above could be moderated by existing marsh condition, future limitation of refuge cover may not be evident in all marshes. It is unlikely that many current marshes have adequate potential to migrate upslope or have sufficient internal habitat complexity to continue provision of tidal refuge in San Francisco Bay in response to increasing tide levels. Projects which conserve existing tidal refuge, and/or create refuge where this limiting factor is suspected may reduce at least the near term risks to affected populations. Particular attention to climate-smart transition zone and high marsh restoration that takes advantage of locally adapted plant species, which tolerate drought and salinity changes, may be necessary to provide adequate cover for California rails in the future.

- Predator management adjacent to marshlands with limited tidal refugia

Anthropogenically subsidized predators, particularly mammalian predators, pose a unique risk in areas wherein tidal refugia is limited to levee faces. Increased predator access and hunting success is thought to be a major reason for low survival rates during the winter. The urban-marsh interface which represents much of the boundary of marshland in San Francisco Bay is a highly productive habitat for predators that are either directly (e.g. feral cat feeding stations), or indirectly (e.g., denning sites for foxes, diurnal refuge for raccoons), supported by the human environment. The impact of mammalian predators may be less in marshes when refuge is dispersed throughout the marsh plain rather than concentrated at the upland edges. Sufficient and appropriate upland transition habitats may also reduce hunting success for some predators. Very dense plantings may preclude access by larger predators (e.g. red foxes), while multiple planting clusters may reduce the likely of predators encountering California rail. Specific characteristics (size or other physical characteristics and/or spatial arrangement or other intrinsic characteristics) of upland transition habitat which promote secure refuge habitats need to be better understood and may be best suited for investigation in an adaptive management framework.

Upland edge transition habitat may remain a poor substitute for dispersed refuge habitat within the marsh plain if mammalian predation of California rails is facilitated by habitat structure and ease of marsh access at the marsh edge. Furthermore, raptor predation appears to be the dominant cause of mortality for California rails (USGS, unpublished data). It remains unclear what quantity or structure of tidal refuge habitat is ultimately needed to increase California rail survival rates.

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## UNCERTAINTY AND KNOWLEDGE GAPS

The characteristics of salt marsh vegetation and surrounding lands that promote survival and reproduction of the California rail are just beginning to be understood. Much more needs to be learned about environmental correlates to California rail demographic rates, predator ecology, and habitat availability to develop more effective management actions that target particular life history stages and processes. Major uncertainties, as outlined above, relate to the habitat conditions which promote survival and successful reproduction by California rails and therefore, the rail's ultimate response to climate change. Habitat needs

are likely to vary in space, due to differences in tidal inundation and constituent plant species, as well as temporally, due to seasonal senescence and tidal inundation. More effective management of California rail populations and habitats will require a broader understanding of not only characteristics of beneficial habitat, but also on how those characteristics can be achieved through restoration design. Direct study of California rail individuals or populations in the San Francisco Bay Estuary has been limited to a few South Bay marshes and many of these investigations were conducted as long as 40 to 50 years ago. Current information on reproductive ecology in particular is urgently needed. For example, the influence of nest site selection on nest success has not been studied since the 1970s and the propensity for California rails to re-nest or double brood like East Coast clapper rails (Kozicky and Schmidt 1949, Blandin 1963) remains unclear.

Population dynamics in the North Bay appear different than those in the South Bay (Liu et al. 2012) and may suggest different factors limiting population growth with potentially different responses to climate change. The importance and degree of connectivity among fragmented salt marshes are also important to understand. Synchrony in population dynamics can destabilize metapopulations and lead to extinction, it remains unknown how the fragmented marshes of San Francisco Bay are connected and how this supports or degrades the species persistence.

Additional validation of California rail abundance and survival rates is needed to inform management options. Much of our recent knowledge on California rails stems from widespread population surveys and radio-telemetry methods. These methods have yet to be validated with independent estimates though considerable theoretical work has been completed which enhances the usability of information from both surveys (Liu et al 2012) and radio-telemetry locations (Overton 2014). With regard to existing population survey information, we can be confident in the direction of population trajectories, but less certain about absolute population size which is important for evaluating progress toward recovery objectives for species delisting (USFWS 2013). This is particularly problematic given the need to contrast marshes or regions (USFWS 2013) where population index values potentially represent very different proportions of the attendant population. Survival rates have been estimated via radio-telemetry studies twice since the early 1990s (Albertson 1995, Overton et al. 2014). Both survey and telemetry-derived information suggest California rail populations have limited dispersal and low survival rates, particularly during the winter. Alternative techniques to assess this information may discover important mechanisms driving population dynamics within the San Francisco Estuary.

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# Science Foundation Chapter 5

## Appendix 5.1 – Case Study

### North American river otter (*Lontra canadensis*)

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#### DESCRIPTION OF THE SPECIES

The North American river otter (*Lontra canadensis*) is a semi-aquatic mustelid endemic to North America north of Mexico (Melquist et al. 2003). River otters were abundant prior to and during the California gold rush (1848–1855) and were harvested for their pelts. By the early 1900s, the North America population had been severely reduced throughout much of its historic range, resulting from overharvest, pollution, and urbanization (Serfass et al. 1993; Larivière and Walton 1998). The California river otter has been part of this large decline. In 1961, the California Fish and Game Commission banned commercial trapping of river otters and granted Protected Furbearer status. California river otter populations likely have increased since being protected (Jameson and Peeters 2004); however, range-wide or San Francisco Estuary Watershed estimates have not occurred.

River otters exhibit sexual dimorphism, with females being smaller than males. They range in length from 900 to 1,300 millimeters and weigh 5 to 14 kilograms (Johnson 2000). Both sexes mature around age 2, but do not become prime breeders until age 4 or 5 (Hooper and Ostenson 1949). In the Sacramento Valley, mating is thought to occur spring through fall (Grinnell 1937), with females commonly gestating annually. Exactly when fertilization and embryonic development begins is unknown (Hooper and Ostenson 1949), and the species exhibits delayed implantation. Gestation lasts 10 to 12 months and typically results in 1 to 5 pups. Life expectancy ranges from 12 to 15 years. Based on life expectancy and gestation length, estimated fecundity ranges from 10 to 50 pups per female over a lifetime. Juvenile to adulthood survival has not been studied.

River otters are opportunistic predators (Mason and Macdonald 1986) that forages near the apex of the trophic pyramid, and thus readily accumulates high levels of pollutants (Clark 1981; Halbrog et al. 1996; Duffy et al. 1993, 1994; Ben-David et al. 2001a, 2001b). Its diet exhibits high seasonal variability. River otters consume 15 to 20 percent (1 to 1.4 kg, or 2 to 3 pounds) of its body weight daily (North American River Otter Husbandry Notebook 2008). On average, one river otter consumes between 365 to 511 kilograms (730 to 1,095 pounds) of prey per year.

Fish are its most important prey (Cote et al. 2008; Penland and Black 2009), especially in winter (Grenfell 1974), and this includes a high composition of salmonids (Modafferi and Yocom 1980; Durbin 1997; Ludwig et al. 2002). On Redwood Creek in California's Humboldt County, fish reportedly were the primary prey in freshwater and estuarine summer diets, occurring in 86 percent of 51 freshwater scats and 81 percent of 36 estuarine scats (Reeves 1988). River otter predation is higher on stocked and hatchery-reared trout than wild trout (Berg and Jørgensen 1991; Yom-Tov et al. 2006), and it is known to decimate hatchery releases (Knudsen and Hale 1968). Planting of introduced fishes (*Oncorhynchus* spp.) and non-native crayfish (*Pacifastacus* spp.) in high mountain lakes (greater than elevation 1,100 meters) likely facilitated the expansion of the river otter's range by supplying new prey sources (Garwood et al. 2013).

River otter also consume reptiles (Manning 1990), birds (Albertson and Evens 2000; LSA Associates 2000), and amphibians, insects, and plants (Grenfell 1974). Clapper rail eggs (Albertson and Evens 2000), terns and gulls (Duffy 1995), and brown pelicans (Salman 2007) are recorded avian prey. In San Francisco Bay near Rodeo, the fall diet was dominated by brown pelicans, with the winter and spring diet composed of fish (Salman 2007). Otters that live near the ocean appear to rely more on rock crabs and nearshore benthic fish; whereas those living near bays and rivers may rely more on other types of fish.

River otter is highly mobile. Movements of more than 4 kilometers per day (km/day) are common, and individuals are known to move up to 42 km/day (Melquist and Hornocker 1983), with females known to disperse 60 to 90 km (Blundell et al. 2002). It requires riverine and/or estuarine, riparian, and even marine habitats to forage, disperse, and raise young. Isolation by distance is the main mechanism for genetic divergence in river otter, in both freshwater and marine ecosystems (Latch et al. 2008; Seymore et al. 2012). Terrestrial habitats, however, are limited to within 30 meters of water, although dens have been reported up to 0.8 km inland (Allen and Mortenson 2011). Dens can be subterranean, in tree stumps, in large woody debris accumulations, or in similar protective habitats that provide cover from predators and weather. Shore-based habitat is influenced by den characteristics, preferably dense brush, riparian, or forest lands to avoid predators. Predators in the San Francisco Bay Area may include coyotes, bobcats, mountain lions, and domestic dogs. Blundell et al. (2002) suggests that predation pressure for river otters is low and not likely to vary seasonally.

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## CRITERIA FOR SELECTION OF THE SPECIES

As an apex predator, river otter is a proven indicator of overall watershed health (Lunnon and Reynolds 1991; Kruuk 1995; Larivière and Walton 1998; Elliot et al. 2008; Black 2009; Mowry 2011) and can be used to monitor the effects of climate change in the Baylands and elsewhere. Its absence in specific areas of watersheds has been associated with a decline in water quality and the presence of chemical contaminants (Mason and Madsen 1993), while its presence has been correlated with undisturbed and unpolluted habitat (Prenda et al. 2001). River otter habitat is vulnerable to aspects of long-term change, especially sea level rise, and to elevated water and ambient temperatures. Its prey also will be affected by these factors, with some species of prey additionally vulnerable to the effects of ocean acidification.

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## OTHER INFORMATION ABOUT THE SPECIES

River otter is found in the Central, South, and San Pablo bays, Carquinez Strait, Suisun Bay, and most tributaries in the Baylands. Both Marin County and the Grizzly Island Wildlife Area report having the largest populations in the San Francisco Estuary Watershed. River otter range and population appear to be increasing in the Baylands (see <http://www.riverotterecology.org/otter-spotter-interactive-map-of-bay-area-sightings.html>). The species inhabits virtually every creek and reservoir, but favors water treatment plants and any location with plentiful juvenile salmonids (SFgate.com 2012). In January 2013, a river otter dubbed “Sutro Sam” began inhabiting the Sutro baths on the beach in San Francisco, being the first river otter seen in the city in decades. Otter crossing signs have been recently posted in Marin County because of incidences of river otters being struck by motorists. Anecdotal observations support the belief that the river otter population in the Baylands has greatly increased and expanded.

## REVIEW OF CLIMATE CHANGE EFFECTS ON THE SPECIES

Evolutionarily, global climate change may affect animals in two opposite ways: those sensitive to heat may have a decrease in body size, as predicted by Bergmann's rule; shorter and warmer winters may save energy on maintenance and increase food availability, and consequently may affect body size (Yom-Tov et al. 2006).

Aquatic mammalian carnivores can inflict major predation effects on fish and invertebrate populations. Mammalian carnivores are endothermic, requiring them to consume far more prey on a per-capita basis than their predatory fish counterparts. Therefore, in addition to their population densities, the individual diet preferences of mammalian carnivores can strongly affect their impacts on particular prey species in any given time and place (Tinker et al. 2008).

Potential regional effects of climate change on river otters include: 1) effects on denning habitat caused by changes in sea level rise and shoreline vegetation; and 2) changes in prey abundance and species composition. River otters are eurythermal, ranging from colder, temperate marine, and mountain streams to warmer lowland streams and estuaries. They appear euryhaline and seem to tolerate a wide range of salinities because marine-dwelling otters have been documented throughout their distribution.

River otters are tethered to habitats near water bodies, and their dens usually occur within 30 meters of water. Consequently, any change to sea level rise has the potential to alter the availability of den habitat upslope. If upslope habitat lacks dense vegetation for denning, then river otters are unlikely to remain in an area. Predator avoidance is one reason for its residing in dense vegetation when onshore.

River otter feed on a wide range of prey, but fish and invertebrates are its dominant food sources. Any change in nearshore fish and invertebrate abundance or composition has the potential to affect otter presence and may cause otters to leave an area, if insufficient food is available to support the otters and their young. Although otters will eat a wide range of prey, high caloric content is important for sustaining gestation and nursing young.

Monitoring the presence of pathogens and trends in pathogen prevalence over time is important in studying the health of an ecosystem and the effects of climate change. Changes in pathogen prevalence and the emergence of new diseases in sea otters (*Enhydra lutra*) have been associated with large-scale stressors in the coastal ecosystem of California, including: increased pathogen presence and runoff into waterways from domestic animal hosts; increased host vulnerability to disease because of prey availability; and diminished water and habitat quality from human development and discharge (Conrad et al. 2005; Johnson et al. 2009).

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## FACTORS THAT MAY AFFECT SPECIES RESILIENCE

River otter resilience is probably low. Small planktivorous fish are a vital link in the nearshore marine and estuarine food chain, and when populations decrease, it affects the fish, birds, and mammals that depend on that species assemblage for food. These small schooling fish species are predicted to decline with changes in climate regionally (see section on estuarine and marine fishes). Sea level rise likely will eliminate or shift den distribution, which will redistribute the river otter population farther upstream into the Delta, along Sacramento and San Joaquin rivers and tributaries.

## LIKELY CLIMATE CHANGE IMPACTS AND RISKS

Changes in the salinity distribution during winter can have a large impact on early survival and subsequent recruitment, on prey such as salmonids, striped bass, Pacific herring, threespine sticklebacks, gobies, and catfish. River otters exhibit strong, individual-level prey specializations (Ben-David et al. 2004; Tinker et al. 2008), yet little is known about the diet of these carnivores in California or the Baylands on either a population or individual level (Grenfell 1974). Nevertheless, river otters are documented to eat a diverse diet and may switch prey when preferred prey are depleted, as likely occurred in Rodeo Lagoon when fish-eating otters switched to eating waterfowl and brown pelicans (Fong, pers. comm., 2006 in Salman 2007). Impacts from climate change most likely will be associated with the redistribution of denning habitat and prey availability.

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## MANAGEMENT ACTIONS TO BE CONSIDERED

Increasing global temperatures correlate with trends of rising sea levels, and a greater variation in climatic events (Patz et al. 2005) could result in further isolation of coastal populations of semiaquatic animals, especially in heavily fragmented and urbanized coastal regions (Seymour et al. 2012) like the San Francisco Estuary Watershed. Sea level rise, habitat degradation, pollution (Guertin et al. 2010), and emergent diseases (Gaydos et al. 2007) may increase loss of genetic diversity.

Management actions for river otters should be based on current knowledge of distribution, population estimates, age structure, and sex ratio. Based on the scant information available, the following management actions are recommended in the San Francisco Estuary Watershed:

1. Develop a current distribution map to establish baseline conditions, using best available science, citizen monitoring groups, phone outreach efforts, and ground-truthing.
2. Assess population genetic structure, connectivity, and genetic effective population size, using standard methods. Is the recent increase in river otter population observed in the Baylands related to high juvenile survival or a shift in distribution resulting from poor upstream habitat and water quality in the Sacramento-San Joaquin rivers and Delta and the low prey to predator ratio? Investigate otter seasonal diet and predatory impact on ESA-listed juvenile Chinook salmon and Delta smelt using scat samples and otoliths from fish prey via strontium isotope analysis. Analysis of otoliths from scat samples could be used to determine predatory impact of river otters on wild versus hatchery-reared juvenile salmonids across spatiotemporal scales.
3. Analyze pathogen (cestodes, trematodes, nematodes, *Toxoplasma gondii*, *Cryptosporidium*, *Vibrio*) composition and abundance from scat samples to determine baseline river otter health and related water quality. Monitor changes in pathogen loading to identify shifts in host populations and exposure (e.g. decrease in *Schistocephalus solidus* associated with a decrease in fish populations).
4. Analyze contaminant (Organochlorine pesticides, PCBs, PBDEs, and PAHs) content from scat samples. Map the distribution of the contaminants to identify potential sources and cleanup sites, and restoration projects. The San Francisco Estuary Project recommended using the river otter in the Bay-Delta system as a watershed sentinel species, especially with regard to contaminates exposure.
5. Analyze fish scales and hair found in scat for total mercury (as a proxy for methyl mercury) and map concentrations across spatiotemporal scales. River otter hair found in scats will estimate total mercury concentrations.

6. Analyze the *Microcystis* presence from scat samples. Map the distribution of the contaminants to identify potential sources and cleanup sites, and restoration projects.
7. Take river otter predation into account when stocking is used as a measure for conserving endangered salmonid populations (Jacobsen and Hansen 1996). Investigate the relationship between the increase in the river otter population in the Baylands and recent changes in hatchery-reared juvenile Chinook salmon release sites in the Baylands.
8. Using EIAs, analyze estradiol, testosterone, and corticosterone metabolites in scat samples to understand the stress and reproductive health of the river otter population. Compare individuals living close to wastewater treatments plants and other discharge sites with those living in rural, undeveloped, and protected watersheds (e.g., Mendocino and Humboldt counties).
9. Consider habitat restoration in areas upslope of habitats where river otters currently occur but are at risk for sea level rise. Restoration may include planting dense vegetation for predator avoidance or addition of large woody debris as potential denning habitat. If suitable habitat is not available upslope, then land acquisition may be an alternative approach for creating habitat.

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## UNCERTAINTY AND KNOWLEDGE GAPS

The size and health of the river otter population is unknown. The distribution of river otters is relatively unknown. They currently are being observed in streams, ponds, and on beaches where they previously have not been observed or have been absent, indicating an increase, shift, or both in population.

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# Science Foundation Chapter 5

## Appendix 5.1 – Case Study

### Shore Birds: Western Sandpipers (*Calidris mauri*) and American Avocets (*Recurvirostra americana*)

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#### DESCRIPTION OF THE SPECIES

**Western sandpipers** (*Calidris mauri*) are small (22-35 g) sexually dimorphic sandpipers, with males typically smaller than females. Western sandpipers frequently occur in mixed species flocks along with other *Calidris* species, including least sandpipers (*Calidris minutilla*) and dunlin (*Calidris alpina*), in the San Francisco Bay Estuary (SFBE) and are the most abundant shorebird species found from fall to spring. Western sandpipers are one of the most common shorebird species in SFBE, the Pacific Flyway, and North America overall. The largest concentrations of western sandpipers, and of most shorebirds, are found in the South Bay where large expanses of mudflats are exposed at low tide and managed and/or salt ponds are available for high tide roosting and feeding habitat (Warnock et al 2002, Warnock and Takekawa 1995). In recent years, there is some evidence that there has been a shift in shorebird numbers towards the North Bay, perhaps as large salt pond restoration projects provide interim mud flat habitat during the evolution towards tidal marsh habitat (Pitkin and Wood, 2011).

Western sandpipers are in the SFBE from late summer to spring, with peak numbers of western sandpipers occurring in April during spring migration (Stenzel et al. 2002, Page et al. 1999), which stresses the importance of managing for increased food resources during this time period as western sandpipers move towards their breeding grounds (U.S. Geological Survey and San Francisco Bay Bird Observatory unpub. data). Although no races or discrete breeding populations of western sandpipers are recognized, genetic differences were found between wintering grounds in Humboldt Bay, California, and South Island, South Carolina (Haig et al. 1997), suggesting isolation on the breeding grounds. Western sandpipers generally lay one clutch (up to 4 eggs) a year, although a replacement clutch may be laid if the first nest is lost early in the nesting season.

Western sandpipers accounted for 60% and 31.5% of shorebirds counted in the South Bay and North Bay, respectively during counts in 2006-2008 (Point Reyes Bird Observatory unpub data). An estimated total population size for western sandpipers is 3.5 million birds (range = 2.8–4.3 million; Bishop et al. 2000, Morrison et al. 2006, Andres et al. 2012). Although there is evidence of a possible population decline (Andres et al. 2012), the magnitude of any change in population size is unknown due to lack of sufficient monitoring in western North America (Fernández et al. 2010).

**American avocets** (*Recurvirostra americana*) are large (275-350 g) shorebirds, with a long recurved bill, long bluish legs and a rusty orange head during the breeding season. Avocets specialize in using ephemeral

wetlands of the arid western United States (Ackerman et al. 2013). Wide-ranging among seasons, avocet migratory movements illustrate the importance of wetland connectivity at a large, landscape scale. They commonly forage in managed ponds and other sparsely vegetated habitats in the SFBE and nest along levees and on islands within ponds as well as in diked marshes. Avocets have a strong tendency to nest with other pairs and often nest with Forster's terns (*Sterna forsteri*). Avocets have a unique foraging mechanism of scything: sweeping bill across the surface of water and mudflat. They generally feed in water <20 cm deep, but can swim and forage in deeper water (Ackerman et al. 2013).

There are approximately 450,000 American avocets in North America (Brown et al. 2001, Morrison et al. 2006). Avocets are migratory, but SFBE is used as both a wintering and breeding site, albeit with different population sizes. Avocets are one of only three shorebird species that nest in the SFBE and are the most common of the three. Approximately 23,000 avocets winter in San Francisco Bay and about 4,000 breed in the estuary (Stenzel et al. 2002). In particular, approximately 2,765 avocets breed in the South Bay, a number that has remained stable for the past 20-30 years (Rintoul et al. 2003). Similar to other waterbirds breeding in SFBE, avocets were uncommon in the early 1900s but now are common breeders especially within former salt evaporation ponds.

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## CRITERIA FOR SELECTION OF THE SPECIES

Western sandpipers were chosen because of their abundance in the estuary, and because they are representative of the larger guild of small shorebirds (*Calidris* spp.) for which SFBE is extremely important as an overwintering and stopover site. This is a relatively well understood species that uses a limited number of sites and has a tendency to aggregate during migration and winter likely due to food availability and as a safety mechanism against predators. Western sandpipers have a limited reproductive output, making them more vulnerable to climate change effects.

American avocets were chosen due to the importance of the estuary as breeding grounds for this large shorebird. They are also representative of the medium-large shorebird guild within the SFBE, and their breeding success is monitored relatively well. Avocets use ponds and diked marshes more than many other shorebirds, common landscape features especially in the South Bay.

Both species of shorebirds are at relatively high risk in the SFBE due to potential for loss of foraging (mud flats, diked marsh) and high tide roosting habitat due to flooding and storm surge from sea level rise and marsh migration.

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## REVIEW OF CLIMATE CHANGE EFFECTS ON THE SPECIES

Western sandpiper migrations are closely tied to seasonally abundant food resources and tidal regimes (Warnock et al. 2002a, 2004, Bishop et al. 2004, Fernández et al. 2010). In addition, the interaction of food availability and the arctic snowmelt determine the feasibility of many species' migration strategies, and ultimately, their breeding success outcomes (Galbraith 2002, Meltofte et al. 2007). Phenological mismatch may be occurring in some species of migratory birds already, where timing of breeding and peak food availability no longer coincide (Møller et al. 2008, Senner 2012).

Extra fat reserves can enhance egg size, chick size, and chick survivorship in pectoral sandpipers (*Calidris melanotos*) and indicates the importance of high quality stopover habitat to fitness of migratory shorebirds (Farmer and Wiens 1999). Intertidal mudflats at one SFBE study site were already at carrying capacity for small shorebirds during spring migration, and shallow water ponds are already critical to subsidizing shorebird migration (Rowan et al 2011). Mudflat loss due to sea level rise (Stralberg et al. 2011) will impact the number of individuals that can be supported in the estuary during this critical stage. South Bay could lose up to 70% of their intertidal feeding habitats by 2100; North Bay up to 39%. Changes in ocean chemistry, coupled with temperature and precipitation changes may alter prey availability for shorebirds (Galbraith et al. 2002).

American avocets are also at risk for the reasons mentioned above. Due to their apparent preference for ponds (Ackerman et al. 2007), avocets may be at increased risk of habitat loss due to the creation of transition zone habitat and the “horizontal levee” proposed to mitigate sea level rise impacts (ESA PWA 2013).

The impact of severe storms on coastal bird populations can be especially significant with birds killed and habitat destroyed, including the erosion of mudflats (Michener et al. 1997). The effects of rapid climate change, including sea level rise and an increase in severe storm events, may adversely affect conditions in sites used for all life stages of shorebirds. Climatic cycles such as El Niño/Southern Oscillation, may impact the abundance, population structure, survival, and premigratory mass gain of some species (Fernández et al. 2010).

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## OTHER STRESSORS

Life history characteristics of western sandpipers that may exacerbate climate change effects include: 1) aggregation of large proportions of the population in a limited number of sites during migration and on the wintering grounds so that changes to food availability could affect a large quantity of the population at once; 2) limited reproductive output that is subject to the various weather and predator cycles in the Arctic suggests a slow recovery from population declines (Fernández et al. 2010); and 3) although long-distance migration patterns allows them to search broadly for food, western sandpipers also must prevail over changes of large geographic and climatic scale and adapt migration decisions accordingly.

Avocet nest survival (37%) and egg hatching success (93%) in San Francisco Bay are lower than might be expected, especially for a predominantly island-nesting species (Ackerman and Herzog 2012), and chick survival (9%) is very low (Ackerman et al. 2006, Ackerman et al. 2014). Low reproductive success is largely due to predation, with a high proportion of eggs and chicks taken by California Gulls (*Larus californicus*). The California Gull breeding population in San Francisco Bay has increased from fewer than 200 gulls in 1982 to over 52,000 in 2012 (Ackerman et al. 2006, 2009, 2010; SFBBO, unpublished). Flooding of avocet nests is also an issue causing reproductive failure in both managed and tidal wetlands. Methylmercury contamination is also a stressor for avocets, and sublethal effects of mercury on avocets have been detected (Eagles-Smith et al. 2009).

Loss of mudflat habitat to the non-native cordgrass (*Spartina alterniflora*) invasion in the SFBE could impact shorebirds on the Pacific Flyway route by converting expanses of mudflats to cordgrass habitat, and reducing the area available for foraging (Stralberg et al. 2004). The Estuary-wide decrease in sediment supply from historic levels (Schoellhamer 2011), the inability of habitats to migrate due to municipal

development and coastal protection structures, and the high level of subsidence in the South Bay will exacerbate effects of sea level rise. The combined effects of habitat change on bird's breeding areas and intertidal habitat loss at both wintering and migratory staging sites could have even more severe effects than could be brought about by any single factor (Fernández et al. 2010).

Additionally, the South Bay Salt Pond Restoration Project is in the process of converting 50% to 90% of the former salt evaporation ponds into tidal marsh habitat. However, both Western sandpipers and American avocets rely on this managed pond habitat heavily for roosting, foraging, and breeding (Warnock and Takekawa 1995, Warnock et al 2002, Stralberg et al. 2003, 2009, Ackerman et al. 2007). Combined with the loss of intertidal mudflat foraging habitat (Stralberg et al. 2011), this loss of foraging areas may be significant.

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## **FACTORS THAT MAY AFFECT SPECIES RESILIENCE**

Changes in migration patterns have been detected for some sandpiper species, and migration pattern changes are related to both local and hemispheric climatic conditions (Anthes 2004). As sea level rises, tidal marshes (especially newly restored tidal marshes) may not have enough sediment supply to keep pace with rising water levels. In this case, more shallow water and mudflat areas than tidal marsh may be maintained in the estuary if the elevation is too low for marsh plant establishment (Stralberg et al. 2011).

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## **LIKELY CLIMATE CHANGE IMPACTS AND RISKS**

- Loss of foraging habitat as mudflats are covered by deeper water levels or eroded
- Loss of habitat and of individuals due to storm events
- Change in prey base and availability due to changes in ocean chemistry
- Loss of high quality migratory, wintering, and breeding habitat could negatively affect fitness of shorebirds

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## **MANAGEMENT ACTIONS TO BE CONSIDERED**

To increase the foraging area available for western sandpipers, low to moderate salinity ponds should be managed as shallow water habitat (<4-6 inches). Overly the long term, restoration of ponds to tidal marsh should be balanced with the availability of this shallow open water. Newly restored ponds function as tidal flats for a number of years, but do not offer foraging habitat for small shorebirds once vegetation has colonized the area. Balancing this with the needs of threatened and endangered species that require mature tidal marsh habitat should be considered.

In particular, an especially critical period for Western Sandpipers is during spring migration (March-April) when birds feed on brine flies and tidal flat invertebrates to help quickly gain body fat. During spring migration, average body mass of Western Sandpipers is higher at San Francisco Bay sites than any other site between Mexico and Alaska (Williams et al. 2007), suggesting that food availability in San Francisco Bay is especially important to Western Sandpiper's overall migration strategy (N. Warnock, personal communication).

For American avocets, near term management should focus on enhancing reproductive success through nest survival and chick survival. Reducing nest predation rates could be accomplished by increasing island nesting habitats in close proximity to food resources, and away from the growing colonies of California Gulls. Managing water at appropriate levels within diked marshes will provide foraging and nesting habitat for avocets as well.

Ponds could be managed for various depths and salinities to allow foraging, nesting, and roosting habitat for a variety of waterbirds. Roosting and nesting habitat should be protected from human disturbance, especially as this real estate becomes more limited due to sea level rise.

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## UNCERTAINTY AND KNOWLEDGE GAPS

- Determine amount and optimal spatial configuration of ponds/tidal flats necessary to sustain shorebird populations
- Better overall monitoring of the distribution and abundance of shorebirds and their associated habitats
- Better understanding of changes that will occur in the distribution, abundance, and quality of tidal flats (and ponds) in context of climate change
- SFBE-wide understanding of over-winter survival, and the carrying capacity of mudflats: internal and external to ponds and including predicted impacts of climate change
- Understanding the importance of biofilm to shorebird diet
- Better understanding of how to restore for maximum benefits to endangered species while considering no loss of shorebird habitat

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## Science Foundation Chapter 5

### Appendix 5.1 – Case Study

# Salt Marsh Harvest Mouse (*Reithrodontomys raviventris*)

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#### DESCRIPTION OF THE SPECIES

Salt marsh harvest mice (SMHM) (*Reithrodontomys raviventris*) are endangered (at the State and Federal level) because of habitat loss, fragmentation and alteration (Shellhammer, 2000). These mice are endemic to the marshes of the San Francisco Bay and include two subspecies. The Southern subspecies (*R. r. raviventris*) is found in the South San Francisco Bay, the Corte Madera area and Richmond area in the Central Bay; and the Northern subspecies (*R. r. halicoetes*) is found in the Marin Peninsula, as well as San Pablo and Suisun Bays.

In most of their range, SMHM are found in the upper half of tidal salt marshes, i.e. in the middle to high marsh or peripheral halophyte zone. Where it is present the mice may also utilize the ecotone between the marshes and the adjacent uplands as escape cover during high to extremely high tides. In addition, they are also found in some of the relatively rare brackish marshes in the South Bay (Shellhammer, Duke and Orland, 2010).

Population sizes of the two subspecies differ, likely due to differences in available habitat throughout its range. SMHM numbers are low in most marshes throughout the range of the southern subspecies with numerous marshes devoid of mice. Mouse numbers are higher, however, in the more brackish marshes of northern and western Suisun Marsh where Sustaita et al. (2011) reported large populations in both pickleweed (*Salicornia virginica*) areas and areas with mixed halophytes such as fat hen (*Atriplex triangularis*), alkali heath (*Frankenia salina*), Baltic rush (*Juncus balticus*), Olney's threesquare bulrush (*Schoenoplectus americanus*) and other halophytic species. It was reported that mixed vegetation not dominated by pickleweed was often equally productive for mice as pickleweed areas. A positive correlation between density and height of the vegetation in the mixed vegetation areas and mouse numbers was also noted.

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#### CRITERIA FOR SELECTION OF THE SPECIES

The loss of SMHM habitat is especially severe in the South San Francisco Bay. Most of the tidal salt marshes of the S. F. Bay have lost the upper half of their mid marsh (pickleweed) zones and almost all of their high marsh (peripheral halophyte) zones as well as almost all of their marsh/upland edges or ecotones. The two latter areas can provide escape cover to the mice during the higher and highest high tides. Many marshes in the South Bay are very narrow and movement of mice is reduced or absent (Shellhammer and Duke, 2010). As a result, the distribution of the mouse is becoming increasingly fragmented in the South Bay. However, Sustaita et al. reported that SMHM are numerous in both diked and tidal marshes of the Suisun Marsh with diked marshes supporting more mice than tidal marshes (2011).

This species was declared endangered because its population numbers were small and decreasing throughout most of its range and because of the continued loss, separation and degradation of its habitat and those threats continue in much of its range.

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## REVIEW OF CLIMATE CHANGE EFFECTS ON THE SPECIES

Sea level rise due to climate change will greatly impact this species, especially in the South Bay. In much of the mouse's range marshes are backed by development. Hence there is no areas into which marshes can migrate into. The few areas of undeveloped upland available for new marshes are the Coyote Hills in the South Bay, the Sears Point area in the San Pablo Bay. There are protected areas along the eastern side of the Marin Peninsula, but many of these areas rise steeply from the bay waters. The Suisun Marsh provides a larger opportunity for marsh migration where not barred by freeways and infrastructure.

In most tidal marshes dramatic sea level rise will result in narrower marshes with little to no upper zones or marsh/upland ecotone. Because of their narrowness and increasingly common stretches of poor vegetative cover such marshes will tend to have more, smaller populations of salt marsh harvest mice than do larger and deeper marshes (Shellhammer and Duke, 2010) and experience higher random genetic drift.

Climate change includes scenarios such as warmer air, less rainfall, earlier runoff, and increased frequency of extreme environmental conditions. In the Suisun Marsh SMHM are known to survive high inundation, catastrophic flooding, and fire, with most populations returning to pre-event levels within a year (CDFW unpublished data). However, if high inundation rates occur in areas without tall escape cover or upland refugia then reproduction could be reduced or eliminated. This could be especially devastating in the South Bay where little habitat remains.

While the two subspecies can tolerate drinking sea water the southern subspecies, *raviventris*, has been subject to relatively high and stable salinity levels. The northern subspecies, *halicoetes*, has evolved with a much greater range of salinity (Fisler 1965). There is no documentation on how long SMHM could tolerate the increased salinity. Increased salinity with lower precipitation could also lead to areas devoid of vegetation, or salt scalds, further degrading remaining SMHM habitat as Paddett-Flohr and Isakson (2003) noted that harvest mice key to the salinity levels of pickleweed.

Other potential impacts of sea level rise include changes and shifts in vegetation composition and the overtopping of all intertidal vegetation by higher storm surges. Such severe inundation is likely to increase predation (Johnston, 1957) as well as decrease reproductive success by flooding out nests (Hadaway and Newman, 1971).

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## FACTORS THAT MAY AFFECT SPECIES RESILIENCE

Resilience is heavily dependent on the composition, height, and depth of vegetation (Sustaita et al., 2011) as deep, complex plant structure allows mice to avoid predation and tall, complex plant structure allows them to remain in tidal marshes during most periods of inundation (Smith, 2012). The potential resilience of the marshes of the San Pablo and Suisun Bays are greater than that of the South San Francisco Bay as there are more deeper marshes and broader ecotones in the northern bays. Most of the South Bay marshes have narrow pickleweed (i.e. mid marsh) zones and little to no high marsh or ecotones as they are backed by

dikes. While it is true for much of the entire bay there is less area adjacent to the marshes in the South Bay into which can migrate as sea level rise.

Population resilience can be increased in tidal salt marshes by increasing marsh size, connectivity, and especially marsh completeness, i.e. the expansion or recreation of high marsh and ecotone areas. It will also be improved by more effectively connecting marshes so that genetic exchange is maintained and inbreeding depression is avoided.

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## MANAGEMENT ACTIONS TO BE CONSIDERED

Short term goals for preserving the mouse, i.e. those in the next 20 to 30 years, include restoring more marshes to tidal action such as South Bay Salt Pond Restoration Project has done in the first two decades of this century. Recreating large marshes increases the chances that complex channel systems will develop within them and hence the development of raised overflow berms along their intermediate channels, areas where marsh gumplant (*Grindelia stricta*) can grow and offer small mammals escape cover from high tides. Some of these restored marshes will have sloping upper edges where high marsh and transition zone vegetation can develop although such slopes will be relatively narrow and seldom connected to adjacent uplands because either the marshes are backed by large dikes or there is development and no where for new marshes to develop. Connecting revegetated marshes will allow for larger populations of small mammals, including SMHM, and protect populations from the loss of genetic variability.

It is very important that wherever possible that land adjacent to existing marshes be acquired and protected as the success of SMHM and other small mammals by 2100 will depend on the upslope movement of the tidal marshes over time as sea levels rise. Decreased sediment loads (Cloern and Jassby, 2012), particularly in the South Bay, will likely result in narrowing of marshes. Increased marsh size, complexity, and the possibility for extension landward are critical to mouse survival. Therefore adjacent slopes to areas such as Sears Point in the San Pablo Bay and the Western uplands and Hills in the Suisun Marsh should be purchased and protected. If sea level rise by the end of the century is severe it is likely that most of the remaining habitat will be adjacent to such hilly areas.

Mice will disappear where marshes, and the ecotone areas adjacent to them, cannot move landward with sea level rise. And that will be the case around much of the bay, especially if sea level rise more than three feet. As such marshes compress, become increasingly narrow and become dominated by frequently inundated low marsh vegetation mouse habitat will disappear. In such areas practically the only solution will be to retain the habitat as diked marshes or muted wetlands with the primary dike protecting landward infrastructure being established on the bay side of the diked marshes. Such diked marshes will be more exposed to terrestrial predators that marshes are in the early 2000's and efforts should be made to reduce such predation.

Recommendations from highest to lowest priority are:

- Conduct genetic studies to determine bottlenecks for genetic dispersal and boundaries for each sub-species (2<sup>nd</sup> stage is currently underway by CDFW and UCD).
- Protect high marsh and upland habitats in existing tidal and managed marshes.

- Identify bottlenecks/ infrastructure barriers, especially roads as barriers to upward tidal marsh movement (Hwy 37 in San Pablo Bay; Hwy 4 in Martinez, Pittsburg; Hwy 680 & Hwy 12 in Suisun.
- Provide upland and wetland (diked or muted tidal) connectivity between marshes where existing marshes may be lost due to flooding with sea level rise.
- Protect existing diked/muted tidal wetland where all habitat could be lost (i.e. permanent levees needed to protect infrastructure) (example is DFW property at Eden Landing), and manage as a habitat levee or horizontal levee (the Bay Institute).
- Adjacent to tidal restoration projects maintain diked marshes as interim reserves (USFWS 2009) and evaluate for muted tidal restoration.
- For tidal restoration designs, incorporate wide and gently sloping high marsh and upland bands. Ideally, these areas should include upland edges with sufficient space to accommodate sea level rise; include tidal berms, mounds, habitat levees, or other elevational raising actions.
- Monitor SMHM populations, distribution, and diversity.
- Encourage tall thick stands of vegetation as high tide refugia and cover along upland edges or habitat levees.
- Control invasive plants, and non-native or artificially abundant predators.
- Reduce access of mammalian and non-native predators to tidal and diked marshes.
- Monitor changes in vegetation species and complexity.
- Continue with research to answer species specific questions such as those identified by the USFWS 2009.

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## **UNCERTAINTY AND KNOWLEDGE GAPS**

With climate change, it is certain that the SMHM will need to be protected from possible impacts such as habitat loss. However it is uncertain how SMHM will react to climate change and whether management efforts to protect SMHM habitat will help to preserve the species. In order to protect the mouse and better manage existing populations in the face of climate change, it will be important to continue studying the mouse and increase knowledge about its ecology. In addition, more in-depth analysis of current data sets may help to increase knowledge about how shifts in climate may impact the mouse.

### **Important Data Gaps**

While SMHM have been studied since they were first described in 1908, there are many gaps in knowledge that will be important to fill in order to protect the mouse from emerging threats such as climate change.

Unanswered questions about habitat use by the mouse include their potential use of brackish marshes in the southern San Francisco Bay, principally areas near Alviso freshened by the effluent of the nearby San Jose Water Treatment Control Plant. The marshes south of Drawbridge supported a third as many salt marsh harvest mice than in nearby salt marshes in a mixture of alkali bulrush (*Bolboschoenus maritimus*) and invasive perennial pepperweed (*Lepidium latifolium*), with more captures in bulrush than peppergrass (Shellhammer, Duke and Orland, 2010). Shellhammer (trapping records) trapped no harvest mice in pure pepperweed in the 1960's and 70's so the question of what percentage mixture of pepperweed and alkali bulrush will support harvest mice is also unanswered. Another uncertainty is whether areas of alkali bulrush in the South Bay could play a role in the survival of the mouse, e. g. as a refuge while changes are occurring in adjacent salt marshes and whether pepperweed should be eradicated.

It will also be important to understand the impacts of upland predators on the mice in the high marsh and marsh/upland ecotone and how it can be reduced and controlled. In addition, more needs to be known about the conversion of heavily grazed grasslands adjacent to marshes, like those in the Suisun Marsh, to small mammal and other species' use as part of the marsh/upland ecotone.

DFW and UC, Davis investigators are currently looking at SMHM genetics to estimate how recent anthropogenic disturbances have affected the sub-structuring of this species, and identify natural and manmade barriers to gene flow/dispersal. Mitochondrial and microsatellite analyses will allow us to assess the validity of the currently recognized subspecies. It is also hoped that further genetic analysis can contribute habitat use and population dynamics information that may be used to address species and habitat management, climate change response, and species recovery efforts.

Additionally, further research partnered with other species should be conducted. Loss of foraging habitat for waterfowl is a big concern in the Suisun Marsh. Any enhancement for waterfowl foods could also benefit the SMHM. Studies of what SMHM are eating in diked and tidal wetlands should be looked at.

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# Science Foundation Chapter 5

## Appendix 5.1 – Case Study

### Tidal Marsh Song Sparrows

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#### DESCRIPTION OF THE SPECIES

This case study considers three endemic subspecies of Song Sparrow (*Melospiza melodia*): Alameda Song Sparrow (*M. m. pusillula*), San Pablo (also Samuel's) Song Sparrow (*M. m. samuelis*) and Suisun Song Sparrow (*M. m. maxillaris*). All three are California Species of Special Concern (Chan and Spautz 2008, Spautz and Nur 2008a, and Spautz and Nur 2008b) and all are dependent on tidal-marsh habitat, primarily found in fully-tidal marsh, but also in muted tidal marsh, where available. Each subspecies is associated with one or two of the key subregions of the estuary: in Suisun Bay, the Suisun Song Sparrow; in San Pablo Bay, the San Pablo Song Sparrow; in South San Francisco Bay, the Alameda Song Sparrow. Central San Francisco Bay has the San Pablo Song Sparrow in the west (Marin county), but Alameda Song Sparrow in the east (Contra Costa and Alameda counties).

Tidal marsh Song Sparrows are year-round resident, nesting in the marsh plain (“high marsh” and “mid-marsh” regions), with small territories that often line tidal channels (Takekawa et al. 2011). They mainly consume terrestrial invertebrates (Grenier 2004). The subspecies appear to demonstrate low levels of dispersal outside the breeding season, as well as limited movement during the breeding season (Johnston 1956, PRBO unpublished).

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#### CRITERIA FOR SELECTION OF THE SPECIES

Tidal marsh Song Sparrows have been well studied since 1996 by PRBO (now “Point Blue Conservation Science”; Nur et al. 1997; Spautz et al. 2006; Greenberg et al. 2006, Stralberg et al. 2010) and others (Nordby et al. 2009). Two recent studies have focused on anticipated impacts of climate change on tidal marsh Song Sparrows: Veloz et al. 2012 and Nur et al. 2012, which findings we summarize here. The three subspecies are of high conservation concern and are considered highly vulnerable to climate change due to (i) anticipated impacts on their habitat (tidal marsh) and (ii) direct effects of flooding on nest success, as elaborated below. Finally, tidal marsh Song Sparrows are indicative of several tidal-marsh-inhabiting passerines in the San Francisco Estuary, in particular the Salt Marsh Common Yellowthroat (*Geothlypis trichas sinuosa*). The latter is also a California Species of Special Concern (Gardali and Evens 2008). Management recommendations that benefit tidal marsh Song Sparrows are likely to apply to the Salt Marsh Common Yellowthroat.

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#### OTHER INFORMATION ABOUT THE SPECIES

Since 1996, the San Pablo and Suisun Song Sparrows have demonstrated population declines; abundance of the Alameda Song Sparrow increased prior to 2002, but has decreased more recently (Wood et al. 2009).

Nur et al. (2012) attribute recent declines to low reproductive success, as a result of nest predation and nest inundation (see also Nordby et al. 2009).

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## REVIEW OF CLIMATE CHANGE EFFECTS ON THE GUILD

Two key impacts that have been assessed are changes in habitat, and nest failure due to flooding.

The availability of tidal marsh habitat and specific characteristics of tidal marsh habitat, including salinity, are expected to change in the short term (2030), mid-term (2050) and long-term (2100). Considering the two “bookend” scenarios (BEHGU Scenarios 2 and 3), mid marsh can be expected to decrease by 4% by 2050 and by 94% by 2110 under the scenario of high sea-level rise and low sediment concentration (Scenario 2). Even full and complete restoration by 2110 will still result in 42% loss of mid-marsh. High marsh will be even more severely impacted, though the importance of high marsh for tidal marsh Song Sparrows is not as clear as it is for mid-marsh. Nevertheless, 94% or greater loss of high marsh surely will have a strong impact. Under the most benign scenario, Scenario 3 (low sea-level rise, high sediment concentration), mid-marsh is expected to increase by 101% by 2050 and by a cumulative 158% by 2110. Still, high marsh is expected to decrease by 54% by 2050 and by a cumulative 80% by 2110.

Veloz et al. (2012) estimate that as a result of changes in the amount and quality of tidal marsh habitat in the next century, the abundance of tidal marsh Song Sparrows can be expected to change substantially, with the direction and magnitude depending on the scenario. Averaged over the four scenarios they considered, which correspond to Scenarios 1 to 4, Song Sparrow abundance for the entire estuary was predicted to increase 13% from 2010 to 2110.

Whether current tidal marshes will be able to survive intact despite increasing global sea level will depend on whether accretion (organic and non-organic) can keep up with or outpace increasing water levels. A key consideration with regard to changes in habitat will be: as water levels increase, can tidal marshes migrate to areas that are currently supratidal? That will depend on the availability of such areas, as well as on the ability for tidal marsh plant species to migrate as well (i.e., establish themselves in new sites). Dispersal of tidal marsh Song Sparrows is not likely to be an important limiting factor, unless there are specific barriers to dispersal.

In addition, tidal marsh Song Sparrows are subject to direct effects of high water levels. Changes in water level reflect the combined effect of global warming, El Nino events, and storms, as well as other changes in peak flow. Both the changes in mean high water levels and extreme events are a concern. Especially high water levels, whether due to extreme tides or storms, are strongly correlated with nest failure (Nur et al. 2012). Extreme water levels affect a nesting attempt that is underway. However, timing of extreme water levels is also important: the complete nesting cycle requires about 25 days. Whereas, an extreme tide can cause failure of a nesting attempt that is underway, Song Sparrows will renest soon after the failure. However, if another extreme water event occurs within 25 days, then the renesting will also fail, which may lead to reproductive failure for an entire breeding season.

High water events during the winter, which may leave individuals vulnerable to predators, may also be of concern. In any case, inundation of marsh habitat because of major storms and extreme tides will reduce foraging opportunities for tidal marsh Song Sparrows, which may cause energetic stress. The winter storm event scenario (Scenario 5), is therefore of great concern since it projects extreme water levels to be maintained for over two weeks during the latter part of winter.

### Winter Storm Extreme Event Scenario

While winter time climate-change impacts on Song Sparrows have not specifically been studied, studies by Thorne et al. (2014), which focused on Black Rails, provide reason for concern. This case study has specifically considered the BEHGU winter storm extreme event scenario.

- i. Tidal marsh song sparrows will be affected throughout their range in the SF estuary. The winter storm event will inhibit their ability to forage and lead to increased mortality. Increased mortality will result from increased predation, as a consequence of reduction of refugia from high water levels, as well as loss of foraging habitat, which can also lead to increased mortality, directly or indirectly.
- ii. The consequences of a winter storm event could be severe. High mortality during the winter period could result as a direct consequence of the storm. If so, recovery could be difficult, though not impossible. There may be more subtle effects, leading to poorer condition of over-wintering individuals, leading to reduce probability of breeding or reduced reproductive success.
- iii. The timing, duration and sequence will influence the impacts of such an event. Duration may be the most important consideration. If inundation of the habitat were protracted, this would cause a large spike in mortality, both of juveniles and adults. The sequence of events is also important, especially if a second stressor affects these organisms at the same time or close in time (before or after the storm). Astronomical tides are most extreme in January, so a storm that coincides with an astronomical extreme tide will be of greatest concern.

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### OTHER STRESSORS

Nur et al. (2012) identified low reproductive success as a key bottleneck facing tidal marsh song sparrows. Nest success was only about 22%, below the level estimated needed for population sustainability, and below that observed for other species in the tidal marsh. High nest failure was due primarily to nest predation. Field evidence indicates predation from raccoons, other mammals, corvids, and snakes. The effects of climate change on nest predators are not known. In any case, the increasing risk of nest failure associated with high water levels (as a result of climate change) will be exacerbated by the current high levels of nest failure due to predation.

At the same time, Global Climate models predict warmer, drier conditions during the breeding season: such conditions are expected to increase nest survival based on statistical analysis of an 11-year data set. Warmer, drier conditions are expected to shorten the breeding season, and thus reduce the number of breeding attempts per Song Sparrow breeding pair per season, but current models predict that increases in nest survival due to warmer, drier conditions will compensate for the reduction in number of breeding attempts associated with warmer, drier conditions.

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### LIFE CYCLE CONSIDERATIONS

Nur et al. (2012) present results of population dynamic models, including population viability analysis. If current conditions continue, populations are expected to decline by more than 50% by 2050. The expected population trajectory with anticipated climate change depends on the assumptions of sea-level rise: under high sea-level rise, the probability of 80% population decline from 2010 to 2060 is 60%. However, under assumptions of low to moderate sea-level rise, the probability of such a severe decline is much less (1% and 10%, respectively). [To be revised with respect to the final assumptions of SLR, to be used throughout the

Update Project.] Because downscaled climate models for San Francisco Estuary tidal marshes predict warmer, drier conditions, which enhance nest success, population dynamic models predict population increases or relative stability with low to moderate sea-level rise. However, an increase in the magnitude of extreme tides (even if relatively rare) will have strong impacts on populations leading to stronger declines or causing expected population increases to turn into population declines.

The conclusion drawn from the population dynamic model and viability analysis is that population trajectory is sensitive to changes in nest success, which will reflect future climate conditions. However, the sensitivity of other demographic processes to climate conditions (e.g., juvenile and adult survival) needs to be explored.

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## FACTORS THAT MAY AFFECT SPECIES RESILIENCE

Population resilience will be enhanced if reproductive success, especially nest survival is increased (see below for specific suggestions). Maintaining high over-winter survival of juveniles and adults is likely to contribute as well. Reducing mortality due to predation, especially during extreme tides/water levels will enhance resilience.

Maintaining connectivity of habitat will help maintain resilience. Particularly effective may be the establishment of restored tidal marshes near extant tidal marsh, especially marshes that are likely to be “sources” rather than “sinks.”

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## LIKELY CLIMATE CHANGE IMPACTS AND RISKS

Current tidal marshes will surely change between now and 2100, but the magnitude of change expected depends on the climate change scenario. The total amount of tidal marsh habitat may shrink, but, even if it does not, we may expect movement of current tidal marshes into areas that are currently upland (above Extreme High Water levels). Therefore, it is important to maximize production of young and their successful recruitment into the breeding population as adults. Thus, maintaining effective habitat connectivity for tidal marsh Song Sparrows is essential.

Evidence to date points to the importance of nest survival (also referred to as “nest success”) as a current population bottleneck. Current levels of nest success need to be increased despite future pressures from climate change (mediated through flooding of habitat) and predators (specifically, nest-predators and those preying on juveniles and adults).

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## MANAGEMENT ACTIONS TO BE CONSIDERED

Management should focus on two key demographic rates: reproductive success and over-winter survival. One priority is to increase reproductive success (or at least not allow it to decrease). Therefore reducing nest predation rates is recommended. This could be accomplished by reducing predator populations or by reducing access or exposure of nests to nest predators. This will be a challenge because numerous species are thought to prey upon song sparrow nests, including a variety of mammals, birds, and snakes.

**Reducing nest predation due to predators entering from adjacent upland habitat** (natural and developed upland habitat) should be a focus, including predation due to feral cats. **Reducing nest predation will serve to offset nest failure due to flooding**, which may increase (and apparently has already increased) due to climate change. A change in nest predation can have a longer term effect on tidal

marsh song sparrows: with reduction in nest predation, Song Sparrows may in the future start nesting higher in the vegetation, which will allow them to escape nest flooding (Greenberg et al. 2006).

**A second recommendation is to provide or enhance mounds or humps within the marsh, i.e.,** increase topographic relief. Such mounds may reduce the effects of extreme water levels on nest flooding. In addition, such mounds may provide refugia for first-year and adult Song Sparrows from extreme water levels during the winter, which occur when all or most of the marsh plain is inundated (due to effects of extreme tides and storm events; Thorne et al. 2012, Thorne et al. 2013). In other words, **marsh mounds may enhance over-winter survival of tidal marsh Song Sparrows**, as has been suggested for rail species residing in tidal marshes.

During the winter time, reducing predator populations or their access to or preference for tidal marsh habitat should also be explored. Reducing predation by feral cats should be a priority.

#### **Recommendations regarding winter-storm event scenario:**

That such an event will occur at some point in the future seems very likely. For example, Thorne et al. (2013) document two recent such storm events, one in January and one in March (in a different year). They documented extended habitat inundation, though not in a position to determine direct impacts on survival or subsequent reproductive success. Management should focus on anticipation of such events: in particular, by providing refugia from these extreme tides. If extreme water levels can be reduced, or at least the duration of habitat-inundation as a result of such storms, this would help alleviate the stress. Since mortality during or subsequent to the storm event is the primary concern, management that is directed to improving survival rates will help address this stressor.

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## **UNCERTAINTY AND KNOWLEDGE GAPS**

The future of tidal marsh habitat is uncertain with regard to its location, extent, and specific characteristics. Also the nature, or even the presence, of adjacent (non-tidal) habitat in the future is not known. This uncertainty is due to uncertainty regarding climate change projections and decisions regarding land-use (maintenance of levees, conversion of current diked baylands to tidal marsh, maintenance of managed ponds, etc.) There is also substantial uncertainty regarding response of tidal marsh plant species to climate change and due to impacts of invasive species; the distribution and abundance of tidal marsh plants will substantially affect tidal marsh birds. The magnitude, frequency, and timing of extreme water levels is difficult to predict but will have severe consequences for tidal marsh birds. Finally, the demographic response of tidal marsh Song Sparrows to climate change is not well known, especially with regard to survival rates.

### **Important Data Gaps**

Impacts of climate change on survival of first-year birds and adults is not known. Small changes in survival rates can have substantial consequences for population growth (or decline) and population resilience. Therefore an important gap to address is information on environmental influences on survival of first-year and adult tidal marsh Song Sparrows. The more sensitive survival rates are to environmental conditions, the more concern there is that climate change may have deleterious consequences for populations. At the same time, sensitivity of survival rates to the environment suggests that management actions can be effective if they target this parameter.

Tidal marsh habitat is by nature patchy. The ability of tidal marsh birds to colonize newly restored habitat, or re-occupy habitat, needs to be better characterized, and barriers to dispersal need to be identified.

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# Science Foundation Chapter 5

## Appendix 5.1 – Case Study

### Vernal Pools

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#### DESCRIPTION OF THE HABITAT

Vernal pools are among California's most sensitive and rare wetland habitats, with annual cycles of winter and spring inundation followed by summer desiccation (Keeley and Zedler 1998; Keeler-Wolf et al. 1998; Botkin et al. 1991). A diverse array of vernal pool complexes still exists within California grasslands, including some within the San Francisco Baylands (Holland 1998; Goals Project 2000). These seasonally precipitation-filled, shallow depressions are mainly found in more inland areas and are underlain by hardpan, clay pan or bedrock (Keeley and Zedler 1998; Goals Project 2000). Specialist species such as annual plants, crustaceans and other invertebrate taxa and some amphibians have evolved life histories specific to the extreme wet and dry conditions found in vernal pools (Keeley and Zedler 1998, King et al. 1996; Goals Project 2000).

Distribution of Baylands vernal pools has been reduced from their historic range (EcoAtlas 1998), with pools currently remaining north of San Pablo (Sears Point and lower Tolay Creek) and Suisun Bays (Solano County), and in southern San Francisco Bay at the Warm Springs complex in Fremont (Goals Project 2000, P. Baye, I. Loredo pers. com). Besides freshwater vernal pools, the San Francisco Baylands contain seasonal pans that lack the characteristic biota of upland pools because of elevated salinity, where salt diffusion from underlying soils causes them to be slightly brackish (Goals Project 2000).

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#### CRITERIA FOR SELECTION OF THE HABITAT

Vernal pool systems throughout California have been relatively well studied (Keeley and Zedler 1998; Keeler-Wolf et al. 1998, Witham et al 1998), and they represent a threatened, and potentially climate-sensitive wetland type harboring high biodiversity. Specifically, many specialist endemic vernal pool species are naturally rare, and have as part of their life histories adapted to a fluctuating system with very extreme conditions making them vulnerable to changes (Keeley and Zedler 1998; Witham et al. 1998).

In the past century, land use change associated with urban development and agriculture have caused the loss of 90% of historic vernal pool ecosystems in California resulting in the endangerment of many species associated with vernal pools throughout the State, including the San Francisco Baylands (Holland, 1978; Griggs and Jain, 1983; Keeler-Wolf et al., 1998; EcoAtlas 1998). This is mainly due to habitat loss, degradation and associated isolation and genetic erosion of decreasing populations (Elam 1998; Ramp Neale et al. 2006, 2008; Sloop et al. 2011; Gordon et al. 2011; Sloop and Ayres 2012; Aguilar 2012, Sloop et al. 2012). Vernal pools and their specialist species are also rare and threatened in the San Francisco Baylands with current distribution severely reduced from historic (Goals Project 2000). This case study addresses the vulnerability of the few remaining San Francisco Baylands seasonal wetland systems and provides ideas for potential climate adaptation strategies.

## REVIEW OF CLIMATE CHANGE EFFECTS ON THE HABITAT

Vernal pools as non-persistent emergent wetlands could be affected by a number of climate change impacts, including *changing precipitation and temperature* affecting evapo-transpiration rates that change inundation periods. This will influence annual plant, amphibian and invertebrate life cycles and reproductive success. Shallow pools may be lost due to decreased rainfall and increased temperatures, and deeper pools may decrease in size changing the community composition (Botkin et al 1991). Most vernal pool species are adapted to extreme and changing conditions, where boom reproductive years buffer relatively short periods of drought, and large numbers of long-lived invertebrate cysts and annual plant seeds are stored in the vernal pool soil for a decade or more (Nunney 2002, Aguilar 2012). Amphibians are also long-lived enough to survive drought years, also depending on boom years for successful recruitment (Paton and Crouch 2002). However, in case of severe changes in inundation period or phenology of wet and dry cycles for extended periods, over the long-term this could result in the extinction of species from the pools. Soil seed and cyst reserves may be used up as less and less reproductive propagules are deposited each year from decreasing population sizes. Also, if rainfall is not consistent and sufficient in the winter to keep pools inundated, amphibian larvae will not survive long enough to develop to the terrestrial adult stage (Paton and Crouch 2002). Within a vernal pool complex, inundation period is therefore the best indicator of whether a pool will support amphibians or vernal pool tadpole shrimp, for example, and an array of larger deeper pools among smaller and shallower pools is ideal to support a maximum array of varying species, buffering fluctuating conditions.

*Sea level rise* will also have a likely detrimental impact on the remaining San Francisco Baylands vernal pool complexes, especially those located adjacent to the Bay at low elevations (EcoAtlas 1998, I. Loreda pers. com.). The main Baylands vernal pool complexes occur behind levees, and are relict from lower sea levels of a century ago or more (Warm Springs complex, and Suisun - Scalley Road vicinity, Nurse Slough, Denverton, Miens Landing vicinity; P. Baye pers. com.). Vernal pool hydrology alterations that will negatively affect vernal pool systems include: 1) levee failure, introducing more saline water into complexes through overtopping or breaching; 2) increased terrestrial flooding in vernal pool-containing lowlands as an indirect effect of stormwater discharge constraints; and 3) backwater effects of high tides and rising groundwater tables relative to rising sea level (and possibly reversed groundwater gradients via artificial groundwater drawdown; P. Baye pers com). Hydrology will therefore change on both sides of levees when sea level rises, permanently changing the soil conditions, or convert the habitat to tidal marsh or subtidal (Botkin et al 1991, P. Baye, pers. com.). Even if not permanently inundated by brackish or saline water, increased salt intrusion to soils may have permanent effects on pool and pan biotic communities as most vernal pool species are not salt tolerant, with some notable exceptions.

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## FACTORS THAT MAY AFFECT RESILIENCE

Vernal pool specialists can overcome shorter inundation periods caused by changes in temperature and precipitation regimes via current adaptations (soil propagule bank and longevity) and metapopulation dynamics, where larger deeper pools may serve as propagule sources for smaller less resilient pools during boom years. Yet, if inundation regimes are subject to more consistent extreme fluctuations over the long-term (i.e. wet early in season, then drying as no additional rain falls during season, then wet again at end of season due to inconsistent yet extreme storm events) and become more consistently shortened even in larger pools, this will likely spell doom for vernal pool species persistence into the next century.

Dynamics affecting resilience to consider in the near term include the likely loss of adaptive capacity to changes in temperature, precipitation and associated phenology in specialist vernal pool species. This is especially of concern in species with restricted distribution that are isolated from other meta-populations in the area, and/or are surviving in degraded vernal pools, as they may be facing population declines and associated genetic erosion (Reed and Frankham 2003; Aguilar 2012; Sloop et al. 2012). Ultimately, these species may face more near-term extinction unless restoration activities that consider these dynamics in the context of current and future climate change impacts are implemented (Elam, 1998; Foin et al., 1998).

Unless directly managed for persistence in their current low elevation range, Bayland vernal pool complexes will likely be lost to the impacts from sea level rise and associated salinity changes by the end of the century. Some Baylands vernal pools, such as at the Fremont Warm Springs complex are found on natural, slightly saline, alkali soils, and non-native species biomass in these pools may be limited by salinity/alkalinity rather than herbivory. Some Baylands vernal pool plants also occurred in high tidal marsh edges, making them more salt tolerant, including *Lasthenia conjugens*, *L. glabrata* (still present), *Downingia pulchella*, *Castilleja ambigua*, *Plagiobothrys stipitatus*, *P. bracteata*, *Triphysaria versicolor*, *T. eriantha* (P. Baye pers. com.). Therefore, some specialist vernal pool species may be able to withstand at least lower levels of salinity intrusion and persist in more saline soils.

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## LIKELY CLIMATE CHANGE IMPACTS AND RISKS

- Precipitation and temperature extremes and/or changes will affect the timing of annual emergence and seed set success for plants and the completing of annual life cycles of invertebrates (i.e. vernal pool crustaceans) or amphibians (i.e. California tiger salamander) and decreased population sizes over the long-term may cause erosion of adaptive capacity and resilience of vernal pool species.
- Projected changes in climate conditions, hydrology and phenology will likely cause major shifts in community composition and the loss of biodiversity, especially of narrowly specialized endemic species that may decline and ultimately go extinct.
- Many rare vernal pool annual plants have coevolved specialist pollinators (Thorp and Leong 1998). Extreme weather events and changes in phenology may cause the disassociation of pollinators and plants and contribute to increased reproductive failure.
- Depending on the climate change scenario considered, sea-level rise will at least by 2100 likely cause the loss of low elevation vernal pool or pan systems to below-ground salt water intrusion or surface flooding as levees fail or overtop and groundwater and inland flooding regimes change. If not protected by levees, these low elevation systems will eventually undergo complete inundation and/or be replaced by the upward migration of marshlands. This may also affect slightly higher elevation inland vernal pools as sea level rise progresses upward over the coming century.
- Higher salinity levels in vernal pool complexes associated with sea level rise will be detrimental to amphibians (reduced viability of eggs and tadpoles – depends on species) and to most endemic saline-sensitive vernal pool plants and invertebrates, causing loss of biodiversity.

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## MANAGEMENT ACTIONS TO BE CONSIDERED

1. To maintain resilience of the vernal pool biotic communities manage for a *heterogeneous pool system* and *large populations* as much as possible (Gilpin and Soulé 1986). This means focusing on supporting an array of pools of varying depths and sizes that help support meta-population dynamics (Hanski 1994),

non-saline conditions and minimizing non-native competitors within pools and surrounding uplands as much as possible to allow for large population sizes of vernal pool native species.

2. Continue to *protect vernal pool complexes from projected tidal inundation* as sea level rises via maintenance of levees or other anthropogenic means. Also address potential issues regarding rising groundwater, impounded floodwaters with no outlet during high tides
3. *Manage all remnant vernal pools and pans in the San Francisco Baylands as high priority conservation targets.* Conserve, enhance, restore all microhabitats to maintain the highest possible level meta-population dynamics to maximize biodiversity and maintain community resilience to changing inundation patterns that might be negatively influenced by non-native annual competitors.
4. *Utilize grazing as a management tool where appropriate* to minimize negative impacts to native pool communities such as lowered pool inundation periods and competitive exclusion from non-native annual plants (Marty 2005). If implemented, grazing regimes must be appropriately designed utilizing suitable grazers and time lines to minimize potential adverse effects. Grazing may not be appropriate in saline/alkaline pools where non-native plants are naturally inhibited by high alkalinity and grazers may in fact cause more harmful impacts than benefits (P. Baye pers. com.). For example, Warm Springs pools are alkali sink, with uncommon subshrubs (*Suaeda moquinii*, *Arthrocnemum subterminale*), that may be damaged by grazing and soils are clay, very soft when wet and may be poached by cattle trampling (P. Baye pers. com.)
5. *Conduct vernal pool surveys within an adaptive management framework* to gain a better understanding of the status of the system over time, so appropriate management actions can be implemented in the face of projected and realized climate change impacts.
6. *Collect seed material/invertebrate cysts from all extant populations for long-term ex situ storage* and potential future reintroduction to higher elevation natural sites or restoration sites. To maximize adaptive capacity via long-term genetic resilience, genetic diversity collections and *ex situ* storage should occur throughout each reproductive season (early, mid and late season) and over several years.
7. *Protect any remaining vernal pools that are at higher elevations/farther from the Bay.* Target vernal pools in Baylands that extend above 21<sup>st</sup> century sea level rise projections topographically for conservation in concert with sea level rise, and reconnect upland transition zone pool complexes with tidal marsh edges where possible (P. Baye, pers. com.).
8. *Evaluate the possibility of translocation of sensitive species and low elevation pool communities.* For pools that are problematic to conserve *in situ* in the face of sea level rise, preserve some component populations individually by accommodating them in re-expanded modified estuarine-terrestrial transition zones.
9. *Investigate potential for vernal pool creation/restoration on higher elevation grasslands.* If feasible, restore vernal pool habitat by perching depressional pools in constructed clay pans in more upland transition zones (P. Baye, pers. com.).
10. If appropriate, *inoculate depressional clay pan pools in upland transition zone with soil propagules from low elevation pools* (by moving top vernal pool soil layer as appropriate). Incorporate brackish tolerant species reintroductions to terrestrial transition zones with constructed claypan to limit perennial dominance

## **UNCERTAINTY AND KNOWLEDGE GAPS**

- The lack of specific data on San Francisco Baylands vernal pool systems is a major uncertainty in evaluating vulnerability and prioritizing and identifying effective management recommendations.
- The adaptive capacity of highly specialist species living in vernal pool systems is unclear. Since most vernal pool species are already adapted to extremes in inundation and other stochasticities that might affect their persistence and reproductive success in any given year, perhaps they will be able to remain resilient to the changes in temperature and precipitation that are projected.
- The feasibility of vernal pool recreation/restoration in higher elevation transition zones has not been tested, and its potential success is unclear, making these created habitats potential population sinks.

### **Data Gaps**

- San Francisco Baylands vernal pool community composition and distribution. We need a current accounting of the plant and animal species that are found in these pools and determine their conservation status and vulnerability to projected impacts.
- There are no current evaluations of the specific localized threats to these remaining Baylands vernal pool systems, so devising specific management recommendations depend on the attainment of relevant baseline data.

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# Science Foundation Chapter 5

## Appendix 5.1 – Case Study

### Baylands Shrews

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#### DESCRIPTION OF THE SPECIES

Three shrew species may occur in the marshes of the San Francisco Bay Estuary. None of the shrews of the Baylands are listed as threatened or endangered by either the state or the federal wildlife agencies but they are uncommon to rare and have disappeared from many known areas of potential habitat. The Draft Recovery Plan for Tidal Marsh Ecosystems of Northern and Central California (USFWS 2010) lists Suisun shrew (*Sorex ornatus sinuosus*) and salt marsh wandering shrew (*Sorex vagrans halicoetes*) as non-listed species covered by the recovery plan.

The Suisun shrew, one of nine subspecies of ornate shrew, may have originally had a range from Sonoma Creek in Sonoma County to Collinsville in Solano County (Rudd 1955, Brown and Rudd 1981). Other evidence points to Suisun shrews inhabiting only Grizzly Island until San Pablo Bay marshes were diked (Grinnell 1913; WESCO 1986). Today, Suisun shrews inhabit tidal and diked marshes of San Pablo Bay at recently-restored salt ponds near Napa and Devils Sloughs, Fagan Marsh, Tubbs Island and Tubbs Setback, Lower Tolay Creek, Mare Island Naval Shipyard, Sears Point, Sonoma Creek; and Suisun Bay at Rush Ranch (including Cutoff and Suisun Sloughs), Hill Slough, Meins Landing, Grizzly Island, and Cordelia Slough (WESCO 1986; CDFW, CDWR, USGS, USFWS unpublished trapping records; CNDDDB). Hays and Lidicker (2000) noted that while the Suisun shrew was rare in many parts of its range, it may be locally common in locations such as their trapping area in Rush Ranch, where they captured 161 individuals in a 100m X 45m area across two seasons.

Northern ornate shrews (both *S. o. sinuosus* and *S. o. californicus*) are more similar, genetically, to neighboring populations of wandering shrews than they are to ornate shrew subspecies south of Solano County, California (Maldonado et al. 2004). Maldonado et al. (2004) suggests that wandering shrews may have been derived from ornate shrews. Salt marsh wandering shrews had a historic range from Contra Costa County down around the South San Francisco Bay and up to the southern portion of San Francisco County (WESCO 1985).

Today, salt marsh wandering shrew populations may be very low. One was captured at Don Edwards NWR in 2006 (USFWS unpublished trapping data), and in the 1980s at Newark Slough, Perry Gun Club, Sulfur Creek, Coyote Creek, and Coyote Hills Slough in Alameda County; and Triangle Marsh in Santa Clara County. Captures were in both tidal and diked marshes (USFWS unpublished trapping data). Shellhammer (unpublished trapping data) noted that after he captured 13 salt marsh wandering shrews in the Triangle Area north of Alviso in thousands of trap nights in the late 1970s, he failed to capture any more shrews in the South Bay marshes over the next 30 or more years. Most of the latter trapping efforts

occurred along the extremely narrow ecotone or juncture between marsh plains and the mixed halophytes along the base and lower portions of steep-sided dikes, the only upper edge available in most of the South Bay marshes. A third shrew, the little known “Fog” shrew (*Sorex vagrans sonomae*) may inhabit the marshes at China Camp State Park on the eastern side of the upper Marin Peninsula (Fisler 1965, Cummings 1975). No captures, however, were noted in 200 trap-night effort by Simons (Shellhammer and Simons 1980) in the most likely shrew habitat in that park. Two *Sorex* sp. were captured at the former Hamilton Air Force Base in Marin County in 1982, but not identified to species (USFWS unpublished trapping data). A fourth shrew species, the California ornate shrew (*Sorex ornatus californicus*), occurs in the adjacent uplands and will not be described here. Assessing presence and numbers of shrews is very difficult. Capture rates are generally low and mortality rates are high when captured in metal box traps used to trap small rodents, even when insects are provided (CDFW, USGS, USFWS unpublished trapping data).

Several studies suggest that these Baylands shrew species occur most often at the marsh-upland ecotone, either between pickleweed (*Sarcocornia pacifica*) marshes and levees vegetated by coyote brush (*Baccharis pilularis*) and grasses (Hadaway and Newman 1971) or between undiked tidal marsh and ungrazed annual grassland (Hays and Lidicker 2000) vegetated by pickleweed, bulrush (*Schoenoplectus* spp.) and cattail (*Typha* spp.; Rudd 1955; CDFW, USGS, USFWS unpublished trapping data). Other vegetation types where captures occur include rush (*Juncus* spp.), gumplant (*Grindelia* sp.), alkali heath (*Frakenia salina*), marsh jaumea (*Jaumea carnosa*), saltgrass (*Distichlis spicata*), arrowgrass (*Triglochin maritima*), perennial pepperweed (*Lepidium latifolium*), and saltmarsh dodder (*Cuscuta salina*) (Hayes and Lidicker 2000; CDFW, USGS, USFWS unpublished trapping data). These shrew species may also be found in areas where the marsh edge is diked but the upland edge is undiked, such as the Hill Slough Wildlife Area in Suisun (CDFW unpublished trapping data).

Rudd suggested that habitat structure may be more important than species composition. These shrews require areas of fairly constant soil moisture with dense, low-lying plant cover in the 5-10 cm range, and abundant invertebrates (Johnston and Rudd 1957; USFWS 2010). They occupy the same middle and high marsh zone habitat as the salt marsh harvest mouse (Williams 1986). Driftwood and organic litter above the high tide inundation zone may be used for nesting and foraging, as well as refugia during warm dry periods (Johnston and Rudd 1957). Woody debris also provides habitat for insects on which they prey (Johnston and Rudd 1957). Nests are constructed above ground in dead vegetation lined with grasses or other soft materials or under woody debris or other cover (Gillihan and Foresman 2004). Suisun shrews excavate or may use existing subterranean burrows as movement corridors and for foraging (Bureau of Reclamation et al. 2011). Suisun shrews use the higher tidal wetland zones and upland transition zones as escape cover from high tides (Bureau of Reclamation et al. 2011). These shrews are constantly active and may consume well over their body weight in food every day (Gillihan and Foresman 2004). *S.v. halicoetes* consumes a variety of food: spiders, crickets, caterpillars, moths, slugs and slug eggs, beetles and beetle larvae, aphids, moth cocoons, maggots, carcasses of small rodents and birds, and fungi, flower parts, seeds, and other vegetation (Gillihan and Foresman 2004).

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## REVIEW OF CLIMATE CHANGE EFFECTS ON THE SPECIES

In one study, Suisun shrews showed a pronounced loss of body weight in winter (30-40%), a condition more common in areas where food becomes scarce in winter (Hays and Lidicker 2000). Here, the author suggests habitat flooding may prevent shrews from foraging sufficiently. Hays and Lidicker (2000) also found evidence of a burrow system in the high tidal marsh, though shrews failed to move into the adjacent

grassland. Shrews may also suffer from weather-induced stress in colder, wetter conditions due to their low heat content and high thermal conductivity (Hays 1990). Though salt marsh wandering shrews and Suisun shrews are capable of swimming (Johnston and Rudd 1957), they die quickly if their pelage becomes saturated in cold conditions (CDFW, USGS, USFWS unpublished trapping records).

One anticipated result of climate change is greater instances of peak flooding events by storms in combination with higher tides (Knowles 2009). Other small tidal marsh rodents (*i.e.*, salt marsh harvest mice [*Reithrodontomys raviventris*]) are known to move up in tall vegetation during high tides (Smith 2012) and also readily move into uplands (CDFW unpublished trapping records, Sustaita et al. 2011). If shrews lack these abilities, they are especially vulnerable to sea level rise, particularly extreme peak water events. According to the Recovery Plan, Suisun Bay, specifically the middle northern (Hill Slough) and northeastern (Nurse Slough) edges, offers more opportunities for restoration to tidal marsh-upland complexes with gentle gradients and room for sea level rise. However, much of this habitat is now grazed grassland. Salt marsh wandering shrews in the South Bay do not even have the option of moving into grasslands, as much of their habitat is bounded by urban development.

Suisun Shrew and particularly salt marsh wandering shrew are rare in San Francisco Bay marshes today because of the loss of their habitat, the ecotone between tidal marsh and adjacent upland vegetation. The survival of any or all of them depends on the protection and expansion of that ecotone. This is especially problematic in the South Bay, where little adjacent upland vegetation exists between Baylands and urban development, with little opportunity for marshes to move upward and landward as sea level rises. Throughout the Estuary, diked ponds restored to tidal marsh may not mature quickly enough to provide this high marsh/upland zone with limited sediment supply and rising sea levels (USFWS 2010). Ironically, even though all the parts of the Estuary have greatly lost sediment supply (Cloern and Jassby 2012), the marshes in the South Bay have the highest sedimentation rates in the Estuary and would more easily keep up with sea level rise than marshes to the north (Knowles 2009).

Even under high sedimentation rate and low sea-level rise rate scenarios, much of the high tidal marsh will be lost over the next 100 years in Suisun and San Pablo (including Marin County) Bay marshes (PRBO model). In the Central Bay, excluding Marin County, the band of high tidal marsh remains similar or may even increase under different sedimentation rate and sea-level rise scenarios. Under all scenarios, South Bay marshes also lose high tidal marsh, with the exception of a segment around Newark Slough and Mowry Slough which may only lose this band under high sea-level rise rate scenarios. However, not much high marsh exists in this area to begin with.

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## OTHER STRESSORS

Mortality rates in *Sorex* species are high (Rose 1994). Fewer than half of all salt marsh wandering shrews live longer than 21 days (Johnston and Rudd 1957). Furthermore, though these shrews are just as vulnerable to extinction as salt marsh harvest mice, they are not protected under the Federal Endangered Species Act (USFWS 2010).

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## INFLUENCES FROM OUTSIDE THE ESTUARY

The San Francisco Bay contains high levels of contaminants, such as petroleum-derived hydrocarbons, heavy metals, pesticides, and PCBs (Monroe and Kelly 1992, Luoma and Cloern 1982). Since shrews have

very high metabolism rates and consume insects which may also concentrate contaminants, they may be especially vulnerable to both lethal and sublethal effects of these contaminants (Pankakoski et al. 1994).

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## MANAGEMENT ACTIONS TO BE CONSIDERED

ecotone is important to prevent the invasion of the more upland form, *S. o. californicus*, and interbreeding between the two subspecies. They suggested that “it may be that provision of adequate upland and marsh habitats permits co-existence of both shrews with minimal contact between them.”

The Draft Recovery Plan, which covers Suisun shrew and salt marsh wandering shrew, identifies general strategies to recover species, including, but not limited to, protecting remaining tidal marsh, restoring tidal marsh, controlling invasive plants, and controlling non-native or artificially abundant predators. Specifically, the Plan calls for directing restoration to benefit species population nuclei, while minimizing impacts. High marsh halophyte zones, marsh-to-terrestrial ecotones, and broad connections to adjacent uplands are specifically outlined. Specific predator control includes removing dike access, and providing high tide refugial habitat isolated from dikes within large marshes. The Plan also suggests creating “long, gentle gradients” to accommodate sea level rise. This can be accomplished at Rush Ranch, where Solano Land Trust owns both tidal marsh and the adjacent grasslands. Elsewhere, this is not easily accomplished. Where tidal marsh is unable to expand with sea level rise, and levees must be maintained, “habitat” (levees with wide bases and gentle slopes) or “horizontal” (levees with an upland ecotone slope of moist grasslands and brackish marshes landward of the existing tidal marsh; ESA/PWA 2013) levees may mimic high marsh bands.

WESCO (1986) estimated that 92% of the wetlands in historic range of the salt marsh wandering shrew have been lost and identified 15 locations throughout its former range that might potentially support populations. These areas should be evaluated for potential reintroduction of this species.

Recommendations from highest to lowest priority are listed here:

- For tidal restoration designs, incorporate wide and gently sloping high marsh and upland bands. Ideally, these areas should include upland edges with sufficient space to accommodate sea level rise.
- For managed marsh enhancement designs, incorporate wide “habitat” or “horizontal” levees, or, where possible, eliminate levees altogether at the upland edge.
- Protect high marsh and upland habitats in existing tidal and managed marshes.
- Reduce or prevent terrestrial predators from gaining access to the marshes in which shrews are found or might be introduced.
- Identify areas around the Bay where shrews can potentially be introduced or reintroduced. These areas should include upland edges with sufficient space to accommodate sea level rise.
- If safe and effective trapping methods can be employed, conduct live-capture surveys throughout the range of Bayland shrews to determine current population centers in order to prioritize restoration projects.
- Control invasive plants, and non-native or artificially abundant predators.

## UNCERTAINTY AND KNOWLEDGE GAPS

We do not know the potential for interbreeding between the Suisun shrew and the California subspecies of *Sorex ornatus*. This potential is described by Hays and Lidicker (2000). They are the only investigators who have trapped a sizeable population of these shrews over a long period of time and they failed to detect any movements of shrews into or out of adjacent grasslands. According to Hays and Lidicker (2000), “this emphasizes what appear to be the unique behavior and physiology of the Suisun shrew, and suggests that interbreeding with the subspecies *S. o. californicus* may be caused by invasion of the marsh by this upland form, and not the reverse. It may be that provision of adequate upland and marsh habitats permits the co-existence of both shrews with little contact between them.”

To add to our lack of understanding of these shrew species, survey methods which may tell us more are particularly problematic. Hays (1998) created custom traps to capture these shrew species with minimal mortality. However, most researchers use metal traps or pitfall traps which are set before nightfall and checked the next morning. Even with the addition of bedding and insects, about half of all shrews captured by these methods die (CDFW, USGS, USFWS unpublished trapping records). Added to problematic capture methods is the limitation of unreliable species identification (Harding 2000).

Research needs to be conducted on bioaccumulation and effects of toxic estuarine contaminants on fecundity and viability of shrews. The results of the research should be applied to water quality standards for the San Francisco Bay Estuary (USFWS 2010).

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# Science Foundation Chapter 5

## Appendix 5.1 – Case Study

### California Black Rail (*Laterallus jamaicensis corturniculus*)

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#### DESCRIPTION OF THE SPECIES

The Black Rail is the smallest member of the avian family *Rallidae* and has a wide-ranging but highly scattered distribution throughout the New World. Of five subspecies, two occur in North America—the Eastern Black Rail (*L.j. jamaicensis*) and the California Black Rail (*L.j. corturniculus*). Throughout its range, the Black Rail is a secretive inhabitant of tidal and freshwater wetlands and rarely ventures out from the cover of dense marsh vegetation. It is more likely to be heard than seen; spontaneous vocalizations tend to be concentrated in the nesting season and are much less common during the rest of the year.

The California Black Rail (CBR), the smaller of the two North American subspecies, has a thinner bill and, unlike the nominate subspecies, is largely non-migratory although capable of sporadic dispersal. Because of its small size (10-15 cm in length; ~30 grams) and furtive behavior, CBR is often referred to as “mouse-like.” It is a rather plump, short-necked bird with a stubby bill, longish legs, and large feet. The plumage is slate gray to black, with white barring on the flanks, white speckling on the back, and a chestnut nape and upper mantle. Males and females are alike as are juveniles and adults, although adults have fire engine red eyes, whereas the eyes of immature birds are brown.

The tidal wetlands of San Francisco Bay estuary and the Delta support the preponderance of the population (Manolis 1978, Evens *et al.* 1991, Evens and Nur 2002). Abundance estimates for the estuary range from 12,400 individuals (Veloz *et al.* 2012) to 14,500 individuals (Evens and Nur 2002). Birds also occur in coastal tidal marshes scattered along the outer central California coast (Bodega Bay south to Morro Bay), in freshwater seeps and swales in the low western Sierra foothills, along the lower Colorado River and Salton Sea at the California-Arizona border, and in northwestern Baja California (Evens *et al.* 1991, Eddleman *et al.* 1994, Aigner *et al.* 1995, Conway and Sulzman 2007, Richmond *et al.* 2008). Coastal marshes of southern California no longer support breeding populations of CBR, but did historically (Garret and Dunn 1981, Unitt 1984, Lentz 2005).

In the San Francisco Bay estuary, the CBR is associated primarily with emergent tidal marshes, most abundantly within the “pickleweed zone,” at or above mean tide level (MTL) and often associated with patches of fine-stemmed bulrush (*Schoenoplectus* spp.). Relatively few nests have been reported (Spautz and Nur 2002), but presence and territoriality during the nesting season (March-June) suggest breeding throughout the tidal marshlands of San Pablo Bay, Carquinez Strait, and Suisun Bay. Nesting is apparently limited to small and discrete habitat patches in the Delta, Central and South bay marshes.

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## CRITERIA FOR SELECTION OF THE SPECIES

The CBR, a California threatened species (CDFG 2011), is a member of a suite of special status vertebrate species emblematic of the estuary's tidal marsh habitat— such as the salt-marsh harvest mouse (*Reithrodontomys raviventris*), California Ridgeway's rail (*Rallus obsoletus obsoletus*; formerly, California Clapper Rail), San Pablo California vole (*Microtis californicus sanpabloensis*), Suisun shrew (*Sorex ornatus sinuosus*), and three endemic races of song sparrows (*Melospiza melodia maxillaris*, *M. m. samuelis*, and *M. m. pusillula*). The historical pressures of agriculture, salt production, and urbanization reduced the former tidal marshlands of the San Francisco Bay estuary by an estimated 78 to 85 percent with a concomitant reduction in tidal marsh dependent species (Nichols *et al.* 1986, Evens *et al.* 1991, Goals Project 1999, Albertson and Evens 2000, Shellhammer 2000).

Conservation concern for the CBR is high because of the linearity and low slope of the preferred habitat, its vulnerability to modification, and its historical loss of preferred habitat (Goals Project 1999). Much of the remaining tidal marsh habitat around the estuary is hardened at the landward edge (e.g., fill, roads, levees, berms, riprap) constraining the marsh plain to a narrow band between upland and bayshore.

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## OTHER INFORMATION ABOUT THE SPECIES

CBR is a sedentary species, establishing and occupying relatively small territories. A radio-telemetry study of birds (n=48) in the Petaluma River found a mean fixed-kernel home range of 0.59 ha, a mean core area of 0.14 ha, and reported average daily movement of 27.6 ( $\pm 1.8$ ) meters and 38.4 ( $\pm 5.5$ ) meters during extreme high tides (Tsao *et al.* 2010). Males have significantly larger home ranges and core areas than females (Tsao *et al.* 2010). Abundance indices (average number of detections per hectare) were used to derive regional estimates of total population size in the San Francisco Bay area from 23 sites surveyed in 1988 and replicated in 1996 (Evens and Nur 2002). Although these methods contained considerable uncertainty, the study derived “adjusted abundance estimates” for each region of San Francisco Bay (Table 1).

CBR habitat preferences within the estuary are well understood. The bird is confined to high tidal marsh (mostly above MTL) with a dense cover (>90%) of native halophytes and moist substrate. In San Pablo Bay marshes, CBRs occur in mature, dense stands of perennial pickleweed (*Sarcocornia pacifica*) often in association with alkali bulrush (*Bolboschoenus maritimus*) and alkali heath (*Frankenia salina*) (Evens *et al.* 1991, Trulio and Evens 2000, Tsao *et al.* 2010). East of the Carquinez Strait, the vertical range and relative abundance of pickleweed decreases from west to east and is supplanted by bulrushes (*Bolboschoenus* and *Schoenoplectus*) and cattails (*Typha*). CBRs are found in these complex associations in Suisun Bay and the lower Delta. Peripheral (fringing) vegetation adjacent to the marsh plain at or above mean higher high water (MHHW) provides important refuge habitat during periods of higher inundation (Evens and Page 1986, Trulio and Evens 2000.) Indeed, CBR presence was positively associated with shorter distances to high-tide refugia in an analysis of CBR-habitat relationships (Tsao *et al.* 2010).

The first California record of CBRs on Southeast Farallon Islands (18 October 1886), 48 kms west of San Francisco, and two or three others from the island in the early 1900s (Ridgeway 1890, Grinnell and Miller 1944, DeSante and Ainley 1980) demonstrate dispersal ability by CBR, a characteristic trait of the *Rallidae* (Taylor 1998). Two dozen coastal California records of birds far from marshes attest further to the

dispersal movements of CBR, especially in the post-breeding season. The timing of these few records hints at irruptive or sporadic nomadic behavior, another trait common to the *Rallidae*. The discovery of CBRs in small isolated freshwater seeps sometimes associated with irrigation ditches and natural fed springs in the Sierra foothills demonstrate their dispersal ability, however seep habitats would have been historic in the Central Valley and could have provided a source for the Sierra foothills populations (Aiger *et al.* 1995, Richmond *et al.* 2008, Girard *et al.* 2010).

**Table 1.** Abundance indices and population estimates of CBR for three geographic regions of the San Francisco Bay area. Abundance index is number of rails detected per effective census area (0.41 ha). Population estimates are shown with and without adjustment for detection probability and amount of suitable habitat (*sensu* Evens and Nur 2002).

Region	Habitat Size (ha)	Mean Abundance Index $\pm$ SE	Mean Abundance Index	Sites (n)	Abundance Estimated based on Median <sup>a</sup>	Adjusted Abundance Estimate <sup>b</sup>
San Pablo bay	531	1.25 $\pm$ 0.345	0.71	13	3930	7100
Suisun & Carquinez	780	1.43 $\pm$ 0.320	1.08	5	4080	7200
Outer coast	43	0.46 $\pm$ 0.196	0.3	5	163	289

<sup>a</sup> Estimated number of California Black Rails per region based on median abundance index only, not adjusted for detection probability.

<sup>b</sup> Estimated number of California Black Rails per region, incorporating detection probability of 0.33 estimated using the program DISTANCE (Buckland *et al.* 1993) and assuming that not all habitat is suitable for Black Rails.

## REVIEW OF CLIMATE CHANGE EFFECTS ON THE GUILD

CBRs and other marsh species are expected to be negatively impacted by both sea-level rise and changes in the intensity and frequency of storms over the near and long-term (Thorne *et al.* 2012, Thorne *et al.* 2013). The dynamics of tidal marsh response to sea-level rise may allow vertical accretion of the marsh elevation (depending on sediment supply) into areas with open space of sufficiently low relief; however, because of the heavily modified landward edge in the “urbanized estuary,” models predict a future net loss of high marsh (Kirwan and Murray 2007, Stralberg *et al.* 2011, Takekawa *et al.* 2012, Thorne 2012). Other climate change impacts to CBR such as changes in precipitation amount and timing, ambient temperature extremes, and changes in water salinity are less well understood (Nur *et al.* 2012).

### Sea-Level Rise

The historical modification of the San Francisco Bay estuary makes it especially susceptible to sea-level rise since there are few areas available for upslope marsh transgression. In addition, many areas are predicted to not maintain their elevation relative to sea-level rise in the later part of the century due to low accretion rates (Stralberg *et al.* 2011, Takekawa *et al.* 2012, Veloz *et al.* 2013). Suspended sediment and measured accretion rates vary from Suisun Bay and around the San Francisco Bay estuary (Stralberg *et al.* 2011, Takekawa *et al.* 2012, Swanson *et al.* 2013, Thorne *et al.* 2013, Veloz *et al.* 2013). For example, accretion rates are relatively high in south San Francisco Bay (Callaway *et al.* 2012), and those marshes are projected to withstand sea-level rise longer with areas transitioning from high to low marsh vegetation by 2100 (Swanson *et al.* 2013). However, north San Francisco Bay marshes, such as San Pablo Bay, Napa River and

Petaluma River marshes, were projected to maintain high- and mid-marsh habitats dominated by pickleweed (and potentially supporting CBR populations) to 2060, but these marshes were lost by 2100 (Takekawa et al. 2012, Swanson et al. 2013). Projected loss of pickleweed-dominated habitats by mid-century in the northern part of San Francisco Bay estuary will negatively affect the CBR and other high marsh species. Areas where natural marsh building processes are not occurring due to subsidence or other human modifications will lose pickleweed-dominated habitats earlier this century (Thorne 2012, Thorne et al. 2014), impacting those CBR populations sooner. For example, Thorne (2012) found that many areas of San Pablo Bay National Wildlife Refuge (NWR) are subsiding and are not keeping pace with current rates of sea-level rise, which will result in the loss of pickleweed-dominated habitat by 2040-2060. Scenarios of sea-level rise in the San Francisco Bay estuary are fraught with uncertainty, but modeling predicts likely impacts that should be incorporated into conservation and restoration planning (Veloz et al. 2013). Understanding how these marshes will respond to sea-level rise are key information gaps for management of high marsh species that include the CBR.

### Storms

Global climate models (GCMs) have yet to converge on future scenarios for storms (Bengtsson et al. 2006; Stephens 2011). However, Sea Level Height (SLH), a metric of extremes, has increased 20-fold in the San Francisco Bay estuary since 1915 (Cayan *et al.* 2008). More frequent and intense El Niño events, as a result of warming in the central Pacific Ocean (Lee and McPhaden 2010) are predicted to bring more storm episodes to the area. In addition to changes in storm frequency, ongoing sea-level rise will create more SLH extremes if land elevation does not keep pace with relative sea-level rise.

Low sea level pressure (SLP) is associated with regional storm events that include high wind speeds and substantial rainfall in the San Francisco Bay and Delta (Bromirski *et al.* 2008) and are capable of raising water levels  $>1 \text{ cm mb}^{-1}$  (Chelton and Davis 1982; Bromirski *et al.* 2003) resulting in a local increase in tidal conditions. Recent studies have documented the importance of Pacific Ocean “atmospheric rivers” (ARs) that transport large amounts of water vapor to the coast and can result in storms and flooding for the San Francisco Bay area (Dettinger 2011, Ralph and Dettinger 2012). In the near-term, the timing of storm events and flooding can put CBR in serious risk of site-level reproductive failure and a decrease in population viability if nest inundation occurs or young birds are at risk. Even single storm events that coincide with naturally high tides (e.g., spring high tides) can pose serious risks to CBR reproduction and survival. Repeated reproductive failure could create population bottlenecks that reduce long-term population viability (van de Pol *et al.* 2010, Nur et al. 2012). In addition, these stochastic storm events can put adults and juveniles at risk of drowning, stress, and predation when sufficient cover habitat is not available.

Recent studies examined the potential impacts of storms on tidal marsh habitats and the CBR in the northern reaches of San Francisco Bay estuary. Two storms occurred in the San Francisco Bay estuary in January 2010 and March 2011 resulting in regional flooding and higher than expected tides. Using water level logging at three San Pablo Bay marsh sites (Petaluma marsh, Coon Island, and San Pablo Bay NWR) it was determined that during the storm episodes the duration of marsh inundation was 2 and 3 times greater than average for that time of year, respectively. At peak storm surges, over 65% in 2010 and 93% in 2011 of the plant communities in these marshes were under water and therefore unavailable to CBRs (Table 2; Thorne *et al.* 2013). Storms during the breeding season, like the one in March 2011, can have negative consequences for CBR and other marsh breeding birds and mammals.

The USGS Western Ecological Research Center assessed marsh inundation levels for CBR home ranges on the Petaluma River in 2005 and 2006. A storm in April of 2006, an El Niño year, increased MHHW during the breeding season sufficiently to flood dominant marsh vegetation species (*Sarcocornia pacifica*, *Bolboschoenus maritimus*, and *Grindelia stricta*) by approximately 20% to 40% (Takekawa *et al.* in press). In addition, the increase of water level in April of 2006 increased the frequency of tide levels above mean CBR nest height and increased risk of nest failure (Spautz and Nur 2002), when compared to the previous year. Population viability modeling indicated that increasing the frequency of extreme storm events from one in 10 years to three in 10 years would result in a Petaluma River marsh population decline of 52% by mid-century. Despite increased inundation of tidal marsh habitat during the stormier breeding season of 2006, home range extent overlapped at Black John Marsh (32%) and at Petaluma Marsh (87%) between the two study years, suggesting that the black rails were not changing their location for nesting based on water levels (Takekawa *et al.* in press). Thus, storms during the CBR incubation period (March-May; Eddleman *et al.* 1994) pose a great threat to established nests (van de Pol *et al.* 2010; Nur *et al.* 2012).

**Table 2.** Percent of marsh vegetation flooded during storms across three marsh sites in San Pablo Bay. During the March 2011 storm, 80-90% of the available habitat was inundated and therefore functionally unavailable for wildlife. The Max SLH was highest at Petaluma Marsh during both storm episodes. During the March 2011 storm, over 90% of the vegetative habitat was underwater during the Max SLH at all sites. Mean higher high water (MHHW) and maximum sea level height (SLH) were determined from water level loggers deployed in 2<sup>nd</sup> order channels (Thorne *et al.* 2013).

Region	Jan-10 % inundated			Mar-11 % inundated		
	MHHW	MHHW	Max SLH	MHHW	MHHW	Max SLH
	Non-Storm	Storm	Storm	Non-Storm	Storm	Storm
Coon Island	41	56	65	7	81	94
Petaluma Marsh	47	74	79	16	93	98
San Pablo Bay NWR	54	65	72	23	90	96

Storm events vary in severity and longevity across the San Francisco Bay estuary with differences in wind, wave action and precipitation amounts. Little is understood about how this varies spatially and how it may effect CBR populations. CBR may alter their behavior during storms, which could include changes in the timing of nesting, home range location, or nest elevation. Such behavioral shifts could make it possible for CBR populations to persist in the face of predicted increases in the frequency of inundation from high water storms.

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## OTHER STRESSORS

The impacts of climate change discussed in this case study pose a significant threat to San Francisco Bay estuary tidal marsh inhabitants. More proximate threats are itemized below.

- Invasive plants are altering the community structure of tidal marshes in San Francisco Bay. Two species in particular—*Spartina alterniflora*, and *Lepidium latifolium*—are of ecological concern because of their ability to alter the vegetative community, to change edaphic conditions, and to displace native plants and animals. *Lepidium* has invaded high marsh zones (especially in Suisun Bay) and non-native *Spartina* has invaded the lower marsh elevations (especially in the central and south bays). Concurrently, several control efforts are underway to arrest or reverse the colonization of tidal marshes by non-native halophytes with a goal of tidal marsh recovery. Continued colonization by these invasive plants poses a threat to CBR to the extent that they might alter high marsh habitat. Other invasive marsh plants are also of concern, e.g., *Atriplex prostrata*, *Apium graveolens*, *Juncus gerardii*, and *Polyypogon monspeliensis* (Grewell *et al.* 2012). Vegetation management projects meant to control these invasive plants also pose a direct stressor to sensitive resident species (Evens *et al.* 2010), including CBR.
- Predation is always a stressor, exacerbated in marshes with reduced vegetative cover, especially along the upland edge (Evens and Page 1986). Nonnative predators—feral cats, black rat (*Rattus rattus*) and Norway rats (*Rattus norvegicus*), and perhaps red fox (*Vulpes vulpes*)—add to the predation pressure posed by native animals such as skunks (*Mephitidae spp.*), raccoons (*Procyon lotor*), northern harriers (*Circus cyaneus*), red-tailed hawks (*Buteo jamaicensis*). Furthermore, at several known sites adjacent to tidal marshes around the urbanized estuary, citizens have established feeding stations for feral cats. These undoubtedly subsidize cats as well as other mesopredators, thereby increasing “background” predation pressure. Also, rip rap and other structures placed adjacent to marshlands provide habitat for rats, which are likely to be egg predators of CBR. Rats are known egg predators of the California Ridgeway’s Rail (Albertson & Evens 2000).
- Intrusion into marshlands by humans, including researchers, has the potential to alter habitat (create trails) and cause mortality of adults and chicks or to disturb nests. CBRs are hesitant to flush, and there have been instances of “take” due to trampling. (See “Walking in the Marsh: methods to increase safety and reduce impacts to wildlife and plants.”  
<http://www.sfbaynerr.org/ctp/documents/1305649171Walking%20In%20the%20Marsh.pdf>)
- Fragmentation of tidal habitat has been an historical stressor on CBR and other marsh-dependent species. The restoration of tidal marshes throughout the San Francisco Bay estuary may be starting to arrest and even reverse this trend, however upland transition zone loss remains a concern.
- Stochastic events—earthquakes, floods, erosion associated with storm surges, and landslides—are ongoing potential stressors.
- Pesticides and other contaminants such as Hg have been found to have increased concentrations over historic levels in the eggs of San Francisco Bay estuary nesting birds (e.g. Lonzarich *et al.* 1992, Ackerman and Eagles-Smith 2007, Tsao *et al.* 2009). Although levels of contaminants in CBR eggs are unknown, it likely mirrors that of other vertebrates that forage in marsh substrate.
- Grazing and other land-use practices on the upland edge of the marsh may reduce peripheral vegetative cover and reduce the quality and availability of high-tide refugia.

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## LIFE CYCLE AND INFLUENCES FROM OUTSIDE THE ESTUARY

The following life cycle information is summarized from Eddleman *et al.* (1994); see also Nur *et al.* (2012) for a summary of demographic information for CBR in the San Francisco Bay estuary. Age at first breeding is presumed to be one year (adult plumage is attained with the preformative molt), and little is known about longevity. In San Francisco Bay, CBRs nest within high elevation tidal marsh. Mean clutch size is reported as  $6.0 \pm 1.3$  SD,  $n=86$ , range 3-8. Known egg-laying phenology spans March 10 to July 6, with an apparent peak in April-May. Second or replacement clutches may account for later nesting. Both sexes incubate.

Chicks are precocial, but remain near the nest site for a day or two after hatching. CBRs are generally sedentary, and show strong site fidelity; however, as in other *Rallidae*, they are capable of dispersal.

Influences from outside the estuary are generally not relevant. However, a recent study using DNA analysis found that populations discovered in the Sierra Nevada Foothills appear linked with San Francisco Bay estuary populations via migration patterns (Girard et al. 2010).

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## **FACTORS THAT MAY AFFECT SPECIES RESILIENCE**

Factors that can increase resilience of CBR populations include maintaining current and increasing habitat extent and increasing connectivity between areas for dispersal. In addition, increasing marsh structural complexity by providing different vegetation types at various elevations for refugia could increase resiliency especially with rising sea levels and increased frequency of storm surges. Reducing adult and juvenile mortality due to predation and drowning will increase resiliency. Recently restored marshes that are adjacent and/or proximate to existing occupied marshes have been colonized by CBRs shortly after revegetation (e.g., Napa-Sonoma marsh “2A”, Sonoma Baylands Wetland Demonstration Site, Carl’s Marsh, Muzzi Marsh, Giacomini Marsh Wetland Restoration Site; J. Evens. Pers. Obs.) illustrating that proximity to established CBR populations could increase resilience. Additionally, restoration of historic marshlands and adjacent slopes for refugia is likely to increase resilience of the species.

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## **LIKELY CLIMATE CHANGE IMPACTS AND RISKS**

Rates of predation and drowning are believed to be greater during high marsh inundation, as animals are forced to move out of the habitat to adjacent uplands (Evens and Page 1986). This may occur more often with the increase of mean sea level and changes in storm frequency or intensity. CBR may be forced to move into sub-optimal habitat where they will be more prone competition, predation, and exposure. The predicted increases in inundation frequency and water depth in the San Francisco Bay estuary over the next century are expected to have negative impacts on the demography of CBR. CBR may be exposed to increased intraspecific competition if they are forced to move into new and occupied habitats.

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## **MANAGEMENT ACTIONS TO BE CONSIDERED**

- Provide upland and vegetated island refugia (either natural or constructed) for high water events from storms to reduce drowning and predation risk
- Restore movement corridors to increase dispersal capability
- Restore and preserve upland open space where marsh migration could occur to increase future potential habitat
- Decrease predation rates on marsh species by removing human structures (posts, poles, barns, rip rap, etc.) that are used as perches in and adjacent to the marsh to improve CBR survival
- Increase the monitoring of wildlife population abundance to assess bay-wide trends and improve estimates of total abundance for the estuary
- Develop detailed landscape-level population viability models to predict the probability of extinction or estimate minimum viable population numbers under different sea-level rise scenarios to inform planning efforts for the species

- Increase understanding of food and foraging requirements and how those may change with changes in inundation frequency and depth to assess impacts to CBR viability.

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## UNCERTAINTY AND KNOWLEDGE GAPS

### Uncertainty

It is uncertain how quickly sea-level rise will occur since it is dependent on atmospheric warming and CO<sub>2</sub> emissions. Marsh accretion rates into the future may vary due to changing sediment availability and plant responses and it is therefore hard to project marsh persistence or drowning and impacts to CBR habitats.

It is uncertain if all marshes will keep pace with sea-level rise at the same rate. Some populations of CBR may be more susceptible to drowning and reproductive failure than others.

More information is needed on the factors that contribute most to reproductive success for CBR, including breeding rates and territory requirements. Little is known about dispersal ability and habitat requirements during the non-breeding season.

### Knowledge Gaps

Other than habitat preferences, information is needed on nearly all aspects of the biology of this species. Primary research needs include the effects of contaminants on mortality and reproduction, effects of inter- and intraspecific competition, and population viability (Eddleman *et al.* 1994). Information is needed about CBR population estimates and trends across the estuary. Specific information on nesting phenology in San Francisco Bay estuary marshes would be helpful in terms of understanding reproductive success. A better understanding of the major factors influencing adult and juvenile survival during high tides and storms is also needed. In addition, little is understood on dispersal timing, distance, and frequency within and out of the San Francisco Bay estuary.

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## Science Foundation Chapter 5

### Appendix 5.1 – Case Study

# Northern toad (*Anaxyrus boreas halophilus*, formally *Bufo boreas halophilus*)

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#### DESCRIPTION OF THE SPECIES

The California toad (*Anaxyrus boreas halophilus*) is one of 3 subspecies of the western toad (*Anaxyrus boreas*). California toads are widespread throughout most of California, and are common in the Bay Area. They require a pond or wetland that retains water for at least 2 months for breeding, as well as upland refugia with ground squirrel or gopher burrows and vegetative cover for foraging and shelter (Mullally 1952). The toads are year-round residents and travel distances averaging 40m daily and annually travel 2.5km (Bartelt et al. 2004).

Amphibians are declining worldwide at an alarming rate for a variety of reasons, such as urbanization, agriculture and associated pesticide drift, disease, UV-B radiation, land-use change and climate change (Davidson et al. 2002, Hof et al. 2011, Alford 2011). They are especially vulnerable to climate change due to their reliance on both aquatic and terrestrial habitats. This case study covers the California toad. The California toad, a subspecies of the well-researched western toad, has not itself been well studied. Western toads have experienced serious declines in Canada, Washington, Oregon and Rocky Mountains for a variety of reasons and may serve as proxy for California toads.

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#### CRITERIA FOR SELECTION OF THE GUILD

Over the past decade, research has been conducted to identify the causes of amphibian decline, such as pathogens and UV Radiation. Pathogens have proven to have a strong negative effect on amphibians world-wide (Hof et al. 2011). While some amphibians have a strong resilience to pathogens, the western toad has a high susceptibility to viruses, bacteria, oomycetes, parasites (trematodes), and fungi (Deguise and Richardson 2009, Kiesecker et al. 2001, Johnson et al. 2001a, Pilliod et al. 2010). Below are two examples of pathogenic effects to western toads.

Amphibian chytridiomycosis (chytrid)(caused by *Batrachochytrium dendrobatidis*, Bd) is a pathogen implicated in amphibian declines and extinctions worldwide (Padgett-Flohr and Hopkins 2010, Blaustein et al. 2012). Bd is found in both ephemeral and perennial ponds and it affects all life stages of various amphibian species differently. Bd infects keratinized cells such as tooth rows and jaw sheaths in larvae, impairing foraging and proper development. Adult frogs may experience impairment of osmoregulation, electrolyte disruption, reduced fitness and mortality due to their keratinized epidermis (Blaustein et al. 2012). In western toads, chytrid causes high mortality and malformations in tadpoles (Blaustein et al. 2005). If the toads survive to

adulthood, they have a stronger resilience to chytrid. Pilliod et al. (2010) observed only a small 5-7% annual decrease in adult western toads survival in diseased populations and concluded that some amphibians may coexist with Bd.

Trematodes enter limb buds and cause limb malformation and mortality in many anurans (Johnson 2001a). Similar to chytrid, interspecific effects of trematodes also varies. Trematodes caused low survivorship and malformations in larval western toads from Oregon (Johnson 2001a). Conversely, of four amphibians surveyed in California, trematode effects on California toads were less than the predicted 0-5% baseline (Johnson 2001b). This contrasted greatly with California newts (*Taricha torosa*) within the same ponds where larvae experienced an abnormality rate of 50% (Johnson 2001b).

UV-B radiation has been implicated in reducing survivorship of adult western toads (Blaustein et al. 2005) and amphibians in general (Bancroft et al. 2007). Bancroft et al. 2007 found that UV-B reduced amphibian survival by 1.9 fold and UV-B affected all life stages and species differently. When additional stressors were factored with UV-B radiation, survivorship decreased even more (Bancroft et al. 2007). For example, increased UV-B radiation increased susceptibility to oomycetes, thereby decreasing survivorship (Kiesecker et al. 2001). UV-B radiation has also been shown to increase the growth of the algae (*Saprolegnia ferax*) which infects communal egg masses such as those laid by the western toads (Kiesecker and Blaustein 1997)

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## REVIEW OF CLIMATE CHANGE EFFECTS ON THE SPECIES

Most ecoregions in the western hemisphere are projected to have at least 10% amphibian fauna turnover (low emissions scenario) and 30% turnover (higher emissions scenario) (Lawler et al. 2010). The San Francisco Bay area is projected to have a 30-50% amphibian species turnover (low to high emissions scenario, respectively) (Lawler et al. 2010). Decreased annual precipitation and increased temperatures, both effects of climate change, have already been observed causing amphibian declines at sites such as Yellowstone National Park (McMenamin et al. 2008).

There are many components of Climate Change that will affect amphibians, such as change in precipitation patterns, increases in temperature, sea level rise, and increases in wildfires. When climate change effects are added to the already lengthy list of threats to amphibians, amphibians appear to have a perilous future.

A decrease in precipitation, rather than temperature may be the largest cause of alarm for amphibians (Araujo et al. 2006). Changes in precipitation may cause earlier and faster drawdown, shortened hydroperiods (the length of time water is in a wetland), reduced number and depth of ponds/pools, increased probability of drying (preventing sufficient time for metamorphosis), alter spatial distribution of suitable and optimal breeding, foraging, dispersal, or overwintering habitats, facilitate pathogens and disease, increase UV-B radiation and advance receding shorelines (increasing exposure to aquatic predators) (Blaustein et al. 2012, Kiesecker et al. 2001, Bay Area Open Space Council 2011, Lee et al. 2015, Amburgey et al. 2012, Araujo et al. 2006). There is also the likelihood of reduction of habitat from conversion of more persistent wetlands to more ephemeral habitats (Lee et al. 2015).

Temperature effects due to climate change are not solely about increasing air temperatures. For amphibians, increased temperatures can effect breeding phenology, development (including sex determination, growth rates and hatching rates), and increase disease and pathogens (Todd et al. 2011,

Blaustein et al. 2001, Blaustein et al. 2010, Blaustein et al. 2012, Bartelt et al. 2010, Eggert 2004, Padgett-Flohr and Hopkins 2010, Piotrowski et al. 2004). It can also alter hydroperiod and water quality, such as dissolved oxygen (Blaustein et al. 2010, Mills and Barnhart 1999). Temperature plays an important role in all aspects of the California toad's life. California toads have a very narrow body temperature range, between 10 C and 25 C (Smits 1984). Subterranean refugia (i.e. burrows) are important for hibernation, but also to insulate toads from extreme temperatures, both hot and cold (Smits 1984, Mullally 1952). Toads use temperature cues, rather than light, to emerge from burrows for diurnal activity (Smits and Crawford 1984) and hibernation (Browne and Paszkowski 2010). Temperature, and partially the moon, also influences spring migration (Arnfield et al. 2012).

Salt water intrusion, causing increased salinity into freshwater systems, as well as inundation and loss of habitat, have been predicted as a result of sea level rise. In general, amphibians are poor osmoregulators. Increased salinity can affect all stages of reproduction and may include decreased size of tadpoles, malformations, delayed development and growth, low hatching success, and decreased survivorship (Karrakar et al. 2010, Gomez-Mestre et al. 2004, Gomez-Mestre and Tejedo 2003, Alexander et al. 2012, Rios-Lopez 2008). However, research, mostly conducted within the past 10 years, has identified an increasing number of amphibians, 100 species and growing, that have evolved a salt tolerance to exploit coastal and inland saline habitats (Hopkins and Brodie 2015). Western toads have an intermediate tolerance to saline environments and can tolerate up to about 12.5% of salt (approximately 400 mOs).” One observed effect of high salinity on western toad is a negative strike response. However the effect was only temporary and the toads improved accuracy as they acclimated to the hypersaline solutions (Dole et al. 1985). Overall, salinity may prove to be a very important factor for toads that live in the Baylands at the salt and freshwater confluence.

Wildfires are predicted to increase due to climate change. Wildfire can cause both direct (mortality) and indirect (habitat change) effects on amphibians and their habitat. In contrast to many of the negative effects of climate change on amphibians, wildfire appear to have a positive or neutral effect on western toads by providing increased open areas for foraging, connectivity and dispersal (Hossack and Pilliod 2011). In addition, western toad occupancy and abundance in most habitats that were burned did not experience a decrease, nor did water temperature of ponds in burned areas influence colonization (Hossack and Corn 2008, Rochester et al. 2010).

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## MANAGEMENT ACTIONS TO BE CONSIDERED

1. Establish buffers for ponds, including connectivity to uplands/terrestrial (ex. migration corridors), and suitable cover/refugia (non-breeding habitat). Western toads, similar to many amphibians, depend on both breeding ponds and upland refugia throughout the year. The area around ponds should include protective cover primarily to prevent desiccation. This terrestrial habitat should include underground burrows, downed wood and stumps for predator avoidance and to conserve body water. It should also include shrub and a mix of shady and open canopy cover (Bartelt et al. 2004). In the case of western toads in Idaho, the recommended buffer zone around breeding ponds should be at least 150 to 200m to protect the majority of toad movement. This is a site specific recommended buffer that includes essential non-breeding habitat. If the non-breeding habitat components mentioned above cannot be found near breeding ponds, then buffers will need to be increased to encompass adequate habitat. Breeding wetlands, migration corridors and non-breeding terrestrial habitat are equally important to the

survival of toads and each requires its own buffer and protection to be determined a case site by site basis (Fellers and Kleeman 2007).

2. Creation, enhancement and restoration of amphibian breeding sites. Heterogeneous temporary breeding sites with a variety of depths and diverse vegetation may increase resilience to increasing temperatures and changing hydrology (Shoo et al. 2011). By providing breeding ponds with both shallow and deep areas, as well as open and shady cover, amphibians at various lifestages can access the microhabitat they require. Plants such as sedges and grasses along shallow edges provide oviposition sites and proper temperatures for egg development (Semlitsch 2002). For example, western toads lay their eggs at a depth between 10 and 15cm (Hammerson 1999). Once the young metamorphose, they can move to deeper, less vegetated water for predator avoidance and thermoregulation (Semlitsch 2002).
3. Pond Location. In addition to current breeding pond locations, consider areas that may provide new opportunities, such as stream and creek mouths, and tidal wetland restoration projects with freshwater components.
4. Manipulation of hydroperiod. In some cases, ponds may need water management for supplemental water or excavation for longer hydroperiod (Shoo et al. 2011). Western toads need a minimum of 2 months of ponding to metamorphose, while other amphibians such as California tiger salamander need at least 3 months (Shaffer and Trenham 2004). Amphibian diversity can be maximized if ponds hold water for a minimum of 2 months and less than 1 year. Allowing ponds to dry annually prevents establishment of predatory fish and bullfrogs and decreases the spread of disease and pathogens (Semlitsch 2000, Semlitsch 2002). Rivers and stream conditions for amphibians may be improved through habitat restoration and modifying flows for breeding sites (Shoo et al. 2011).
5. Refuge assurance. Installation of microclimate and microhabitat refuges such as artificial wetting (ex. portable irrigation sprayers and misters), litter supplementation, artificial shelters (ex. coverboards and downed wood), and change in canopy cover (ex. increase cover by planting vegetation to decrease temperature in ponds) may be needed where other options are limited (Shoo et al. 2011).
6. Manage for multiple species. In general this is a good strategy and target species' life histories and habitat preferences should be considered. In the case of western toads, managing lentic wetlands for both the California tiger salamander and the western toad is not recommended (Bobzien and DiDonato 2007).
7. Grazing. Maintain grazing regimes that support habitat for amphibians, reptiles, and grassland butterfly species. Avoid build-up of thatch and biomass, which degrade grasslands and reduce ground squirrel populations, which provide burrows for amphibians and prey for snakes (Bay Area Open Space Council 2011).
8. Invasive weeds. Control invasive weeds that crowd out native plants and alter vegetation composition and hydrology in native habitats (Bay Area Open Space Council 2011).

- a. Additional metapopulation studies of pond-dwelling species, such as California toads, California red-legged frog, and California tiger salamander are needed to guide landscape-scale pond management, restoration, and creation, and to provide estimates of metapopulation viability.
- b. We suggest conducting comprehensive reptile and amphibian surveys in the Bay Area and the creation of an online repository for this information, similar to California Avian Database Center.
- c. Map the occurrence of chytrid fungus, and estimate its current and potential impact on local amphibians. Minimize spread of the disease by implementing best management practices, such as washing field equipment and boots when conducting pond surveys (Bay Area Open Space Council 2011).
- d. In many cases, California toad data has informally been collected as incidental catch during listed species surveys, for example the comprehensive East Bay Regional Park District surveys (Bobzien, S and JE DiDonato 2007). We suggest compiling this and other California toad survey data to improve the range map to assist with conservation efforts.
- e. Provide an outreach and educational opportunity by creating a Citizen Science project to track toad observations.

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# Science Foundation Chapter 5

## Appendix 5.1 – Case Study

### Dabbling Ducks

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#### DESCRIPTION OF THE SPECIES

Dabbling ducks are the most abundant group of waterfowl that overwinter in the shallow wetlands and ponds of San Francisco Bay (SFB). Species within this group are primarily omnivorous, feeding on both plant material and macroinvertebrate prey by “tipping” to access benthic foods in bottom sediments or by foraging in the water column. Although the majority of these ducks are migratory, smaller numbers also nest locally in the SFB. These species vary in body size, ranging from the large mallard (*Anas platyrhynchos*) to the small green-winged teal (*Anas crecca*). Other dabbling duck species present in the SFB include American wigeon (*Anas americana*), gadwall (*Anas strepera*), cinnamon teal (*Anas cyanoptera*), northern shoveler (*Anas chrypeata*), northern pintail (*Anas acuta*), and wood duck (*Aix sponsa*).

Most of these dabbling duck species are known to breed in the SFB with the exception of the American wigeon and green-winged teal. However, the majority of wintering ducks originate from breeding grounds in the Central Valley of California, Pacific Coast States, Alaska, and Intermountain States/provinces, prairie potholes, and boreal forest. Their wintering populations in SFB are greatest between October and January, and dabbling duck species are found in all SFB regions. However, most dabbling duck use is associated with former salt production ponds and in the north and south bays, and managed wetlands of the Suisun Marsh.

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#### CRITERIA FOR SELECTION OF THE GUILD

The five most numerous dabbling duck species in the SFB during January 2012 included northern shoveler, northern pintail, American wigeon, gadwall, and mallard. Both dabbling ducks and diving ducks include species of special management interest, including northern pintail and lesser scaup (*Aythya affinis*). Waterfowl are popular game birds, and waterfowl hunting is a traditional use of the SFB wetlands and bays (Hall 2011). Waterfowl populations are monitored annually on both breeding and wintering grounds to determine population health and to establish annual hunting regulations. Dabbling ducks primarily use former salt ponds and managed wetlands, which could be vulnerable to impacts of climate change. Sea level rise and more frequent extreme storm events will stress exterior levees of managed ponds, which in the future could require considerable commitment and funds to maintain these habitats. Increased salinity in managed moist-soil wetlands of the Suisun Marsh could reduce wetland productivity and thus dabbling duck carrying capacity (Ackerman et al. 2014).

## **OTHER INFORMATION ABOUT THE GUILD**

Dabbling ducks generally arrive earlier in the SFB than diving ducks, and account for about half the waterfowl in October (Accurso 1992). Peak populations generally occur annually in December and January, and dabbling ducks comprise about 30% of the total duck population in the SFB. For example, the Suisun Marsh hosts many of the dabbling ducks in the San Francisco Bay-Delta. There are currently about 60,000 waterfowl that over-winter in Suisun Marsh and 90% of them are dabbling ducks, 5% are diving ducks, 2% geese, 1% sea ducks, and 1% swans (Ackerman et al. 2014). While many dabbling ducks migrate north to breed in late winter and early spring, California and in particular the SFB-Delta region are unique in that many of the dabbling ducks that over-winter are also locally grown. In particular, mallards, and to a lesser extent gadwall, breed in significant abundance locally such that they contribute substantially to the local population (Ackerman et al. 2014). The Suisun Marsh, because of its proximity to the Delta and lower water salinity than other regions of the SFB, is the most important region for nesting ducks in the SFB (Anderson 1960, McLandress et al. 1996, Ackerman et al. 2014). Although total winter dabbling duck numbers (in January) were constant between 1981 and 2012 in SFB, there were significant population changes for certain species (Richmond et al. 2014). Pintail have demonstrated a severe decline in SFB, and in Suisun Marsh in particular (Ackerman et al. 2014). Specifically, the pintail population in Suisun Marsh went from 235,800 in the 1950s to only 14,000 pintail by the 2000s (Ackerman et al. 2014). This dramatic decline in the pintail population index in Suisun Marsh reflects both the decline in the continent-wide pintail population, and fewer pintail wintering in Suisun Marsh relative to other parts of the state. Mallards have declined slightly in both the SFB and Pacific Flyway. Both gadwall and shovelers have increased substantially in the Pacific Flyway although only gadwall have increased significantly in SFB. There is no significant trend in American wigeon numbers since 1981 in the Pacific Flyway or SFB.

Continentially, dabbling duck populations in general have been increasing over the past decade due to favorable wetlands conditions on the breeding grounds. The continental breeding populations of several species that winter in SFB are well above their long term averages (LTA, 1955-2013), including green-winged teal (LTA+50%), gadwall (+80%), and shoveler (+96%). American wigeon (+2%) and pintail (-17%) are near their LTA, but remain well below the continental population goals established by the North American Waterfowl Management Plan (NAWMP; USFWS 2013). Mallard breeding populations are above the LTA (+36%) and NAWMP goal in mid-continent, but down slightly in western states such as Oregon (-7%) and California (-18%) where most of SFB wintering mallards originate.

Distribution of dabbling ducks in SFB is largely concentrated in the Suisun Marsh, and wetlands adjacent to the North Bay and South Bay (Accurso 1992, Richmond et al. 2014). In the Suisun Marsh, dabbling ducks comprised over 90% of the waterfowl (Ackerman et al. 2014). Whereas, in the main regions of the SFB, dabbling ducks comprised only about 10-30% of the total waterfowl (Accurso 1992, Richmond et al. 2014).

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## **REVIEW OF CLIMATE CHANGE EFFECTS ON THE GUILD**

Within Suisun Marsh, dabbling ducks strongly select managed wetland habitats for foraging at night, and avoid tidal marshes, bays, and sloughs (Ackerman et al. 2014, Coates et al. 2012) as they do in other estuaries (Gordon et al. 1998). Therefore, any loss of diked wetlands and managed wetlands along the

bay's margins could impact local dabbling duck populations. Disturbance is also an important factor for dabbling duck habitat use, and undisturbed habitats provide important roosting and loafing habitats for dabbling ducks during the day (Cassaza et al. 2012). Climate change influences on both local continental temperature and precipitation patterns could have important influences on duck breeding, and ultimately local duck populations. Using two long term dabbling duck nesting datasets in California, including within the SFB-Delta at Suisun Marsh (23 years total), Ackerman et al. (2011) found that nest survival and clutch size declined with mean daily temperatures. Further, egg hatching success declined strongly with extreme temperatures. Rainfall was not consistently correlated with nest survival, but nesting season length was influenced by rainfall amounts during the pre-breeding season.

Finally, increased salinities could have a negative effect on dabbling ducks in two ways. First, duckling survival is known to decline with increased salinity in the brood-rearing habitat. Second, salinity influences the types of plants that can be grown within managed wetlands where the management focus is often food production for dabbling ducks. Higher salinity levels makes it difficult to manage for plant species which tend to be highly productive and provide abundant energy-rich seeds sought by waterfowl (Miller et al. 1975; Rollins 1973, 1981; Burns et al. 2003).

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## **FACTORS THAT MAY AFFECT SPECIES RESILIENCE**

Poor winter body condition can affect species' resilience by making them more susceptible to disease, reducing survival and possibly reproduction.

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## **LIKELY CLIMATE CHANGE IMPACTS AND RISKS**

Depending on a given potential climate change effect, impacts and risks will be different for each species and may differ for breeding and wintering ducks.

Primary impacts:

1. If levees not maintained, loss of shallow habitat (salt ponds, managed wetlands) important to foraging dabbling ducks.
2. Change in invertebrate prey availability due to increases in salinity may change available diet or diet composition.
3. Reduction in desirable food plants due to increased salinity.
4. Potential for reduced nesting success associated with altered temperatures and precipitation patterns due to climate change.
5. Potential for reduced duckling survival due to increased salinity levels.

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## **MANAGEMENT ACTIONS TO BE CONSIDERED**

1. Manage ponds (brackish to freshwater) of various depths and salinities to maximize foraging, nesting, and roosting habitat for waterfowl. Dabbling ducks "tip up" to forage, so food is unavailable in

wetlands deeper than 45 cm (<25 cm preferred). Higher salinity wetlands are harmful to young ducklings and provide less diverse and abundant foods. That is, prevent or reduce high salinity in ponds.

2. In the near term, consider re-enforcing dikes, raising dike crown heights, using large water pumps, and managing soil salinity levels (with leaching strategies) to overcome flooding events and salinity intrusion, such that sufficient diked wetlands be retained.
3. Enhance a portion of former salt ponds to establish managed wetlands. Maintain ponds with appropriate depths, islands, and varying salinities as habitat to benefit migrating, wintering, and breeding dabbling ducks.
4. Manage islands and levees to provide roosting and nesting habitats within managed wetlands, and possibly restored tidal marshes.
5. Evaluate potential use and value of submerged aquatic vegetation to dabbling ducks, to determine whether habitat conversions (to tidal) or other restorations (planting) will provide foraging opportunities for dabbling ducks (e.g. eelgrass [*Zostera marina*], sago pondweed [*Stuckenia pectinata*], widgeongrass [*Ruppia maritima*]).
6. Protect and manage uplands and agricultural lands (especially small grains, grazing/haying), especially in areas adjacent to low salinity wetlands and tributaries, as nesting habitat for locally breeding mallards.
7. Use a collaborative approach to Suisun Marsh restoration and consider the importance of maintaining strong stewardship that duck hunters have provided to the Marsh since the turn of the 20th century. Consider important societal impacts of converting portions of Suisun Marsh back to tidal marsh. Although some conversion is inevitable with changing policies, sea level rise, and the increasing pressure on the Delta's dike system, this may be best accomplished using a well-planned collaborative process.

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## UNCERTAINTY AND KNOWLEDGE GAPS

- It is uncertain how quickly local sea level rise will occur and whether levees will be maintained in salt ponds and managed wetlands.
- Future land use changes and restorations altering current habitats (i.e., conversion of managed ponds to tidal marsh habitats).
- More information is needed about the foraging behaviors (e.g., timing, duration, and movements) and food habits of dabbling ducks in the SFB region (information is available for some species in Suisun).
- Information is needed about the nesting biology of dabbling ducks in the SFB. Much is known about nesting in the Suisun Marsh (Ackerman et al. 2014), but very little is known about nesting densities in the North and South Bay regions of the SFB (where known breeding occurs).

- Salinity is known to reduce duckling survival at high levels, but the relationship between waterfowl brood survival and water salinity in SFB including Suisun Marsh is unknown. Grizzly Island in Suisun Marsh has some of the highest mallard nesting densities and success in North America but little is known about recruitment because fledging survival is not known.
- Little is known about waterfowl food availability in existing salt ponds, managed wetlands, and tidal (and muted tidal) wetlands, relative to management and salinity. This information is needed to estimate carrying capacity and inform future habitat restoration and mitigation efforts.
- Evaluate potential use and value of submerged aquatic vegetation to dabbling ducks, to determine whether habitat conversions (to tidal) or other restorations (planting) will provide foraging opportunities for dabbling ducks (e.g. eelgrass, sago pondweed, widgeongrass).
- Information is needed to inform best management practices (BMPs) to maximize waterfowl food production at varying salinity levels in managed wetlands of the Suisun Marsh and SFB.
- See additional data gaps listed in Ackerman et al. (2014) specifically for the Suisun Marsh area of the SFB (many apply generally to the SFB).

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# Science Foundation Chapter 5

## Appendix 5.1 – Case Study

### Diving Ducks

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#### DESCRIPTION OF THE SPECIES

Diving ducks are the most abundant group of waterfowl that overwinter in the open bays and ponds of San Francisco Bay (SFB). Species within this group are primarily benthivores that dive to obtain their macroinvertebrate prey in bottom sediments, although at times they may eat plant matter or forage in the water column. These migratory species include bay ducks (lesser scaup *Aythya affinis*, greater scaup *A. marila*, canvasback *A. valisineria*), sea ducks (surf scoter *Melanitta perspicillata* and bufflehead *Bucephala albeola*), and a stiff-tailed duck (ruddy duck *Oxyura jamaicensis*). These species vary from largest to smallest body mass: canvasback, greater scaup, surf scoter, lesser scaup, ruddy duck, and bufflehead.

Their breeding grounds range from Central Valley grasslands, intermountain wetlands, prairie potholes, boreal forest, and Arctic tundra. Their wintering populations in SFB are most abundant between October and April, and SFB comprises up to 50% of the number counted during midwinter surveys on the lower Pacific coast. Species are found in all SFB regions, but greater scaup and surf scoter are most often seen in subtidal to intertidal waters and are not commonly found in baylands. In contrast, ruddy duck and bufflehead populations are most abundant in baylands, particularly in managed ponds. Canvasbacks are commonly found at estuaries or creek mouths.

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#### CRITEREA FOR SELECTION OF THE GUILD

SFB is particularly important for diving ducks, because this estuary supports a majority of the diving ducks counted in the lower flyway during the winter. Migratory waterfowl are species of special management interest. Most of the waterfowl are hunted, and the annual regulations are derived to maintain healthy populations. In particular, the San Pablo Bay National Wildlife Refuge was established partially in recognition of the area's importance for canvasbacks. Waterfowl may be adversely affected by human disturbance (e.g., boat traffic) in highly urbanized estuaries such as SFB. In addition, multiple stressors including, but not limited to, climate change, sea-level rise, and extreme storm events are likely to affect the key intertidal and subtidal habitats, such as saline ponds, mud flats, and eelgrass (*Zostera marina*) beds, that are used by this guild.

## OTHER INFORMATION ABOUT THE GUILD

Diving ducks account for up to 75% of the waterfowl in SFB open bays and saltponds, and peak wintering numbers are typically counted in mid-January (Accurso 1992, Takekawa et al. in review). Scaup and surf scoters are the most abundant species wintering in SFB, comprising between 45-47% and 19-20% respectively, of all waterfowl counted. Ruddy duck, canvasback, and bufflehead average 7-8%, 7%, and <2% respectively, of waterfowl counted in SFB (Accurso 1992). More recent counts (2012) indicate that 31% of ducks in SFB are scaup, 10% are ruddy duck, 9% are canvasback, 5% are scoter, and 4% are bufflehead (Richmond et al. 2014).

Continentially, scaup and scoter populations have been declining at an alarming rate for the past several decades. These trends are reflected in SFB midwinter waterfowl counts, particularly for scoters. Long-term wintering and breeding area surveys for scoters suggest they have declined more than 50% continentally in the past 50 years (Hodges et al. 1996, Savard et al. 1998, Sea Duck Joint Venture SDJV 2001, Dickson and Glichrist 2002, Evenson et al. 2005, USFWS 2009). While scaup numbers have climbed in recent years, North American scaup populations are currently > 1 million birds below the North American Waterfowl Management Plan (NAWMP) goal of 6.3 million scaup (USFWS 2012). Canvasback populations are stable continentally, but have declined in SFB from peak numbers of 60,000 in the 1960s to less than 20,000 in the 2000s, possibly reflecting winter habitat change, redistribution to the Central Valley (Kruse et al. 2003), or both. Ruddy duck and bufflehead populations are both thought to be stable or increasing continentally (Gauthier 1993, SDJV 2001, Brua 2002).

Diets of scoter and scaup in SFB are dominated by bivalves, particularly *Corbula amurensis* (De La Cruz 2010, Takekawa et al. unpubl. data), and depletion of this clam (Poulton et al. 2002) may influence movements of scoters and scaup out of San Pablo Bay by mid-winter (Lovvorn et al. 2013). Pacific herring (*Clupea pallasii*) roe is an important prey item for scoters in the Central Bay, comprising as much as 54% of their mid-winter diets based on stable isotope analyses (De La Cruz 2010). Canvasbacks in SFB also consume mainly bivalves, but typically forage on larger clams buried deeper in the sediment than those chosen by scaup and scoters (Takekawa et al. unpubl. data). Ruddy ducks foraging in North Bay ponds displayed dietary flexibility based on pond salinity; consuming amphipods and bivalves in low salinity ponds, and brine flies and seeds in high salinity ponds (Takekawa et al. 2009). Bufflehead diets have not been studied in SFB, but at other coastal wintering areas crustacean, bivalve, and occasional fish prey, including herring roe, are taken (Erskine 1972, Stott and Olson 1973, Bayer 1980).

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## REVIEW OF CLIMATE CHANGE EFFECTS ON THE SPECIES

Recent studies in SFB have indicated that net deposition of sediment may not keep pace with sea-level rise (Ganju and Schoellhamer 2010). Sea-level rise increases water levels and reduces wave-induced sediment redistribution of bottom sediments, especially in the shallowest areas (<2m) that are commonly used by diving ducks. Thus, intertidal flats and baylands may not retain current elevations and become deeper.

Changes in salinity, grain size, and possibly water depth may change the invertebrate community composition and their availability to diving ducks. The effect of these changes on primary productivity and associated abundance and distribution of macroinvertebrate prey is not known, but any changes to these main prey items for diving ducks would certainly have some influence.

Sea-level rise (SLR) can decrease light irradiance, but with decreased suspended sediment, optical depth may not greatly differ as a consequence of sea level rise. Increasing frequency of extreme storm events (Graham and Diaz 2001) may result in less favorable foraging conditions for diving ducks. Although diving ducks may forage at night and sediments are typically stirred up by their bottom-foraging activities, they may use their vision to find prey resources. In addition, the timing and extent of phytoplankton blooms may change with changes in light levels on the shoals, influencing bivalve filter feeders upon which diving ducks feed. Increasing water clarity may be beneficial to eelgrass, which is a herring spawning substrate and provides habitat for some invertebrate species consumed by diving ducks (Anderson et al. 2008)

Changes in oceanic conditions may drive distributions of sea ducks, potentially causing shifts in wintering ranges and changes in overall numbers wintering in the estuary. For example, distributions of scoters and other sea ducks on the Atlantic coast have been shown to be related to ocean conditions such as the North Atlantic Oscillation and sea surface temperature, with scoters distributed closer to shore and in estuaries during colder, more severe winters (Zipkin et al. 2010). Additionally, trophic dynamics in SFB are coupled with the California Current System (CCS) such that changes in cold-water upwelling along the Pacific coast can lead to large increases in cold water fishes in SFB (Cloern et al. 2007) that are strong competitors with diving ducks for their bivalve prey.

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## **OTHER STRESSORS**

- While wintering in the estuary, diving ducks face a number of potential threats, many of which are associated with urbanization. Potential threats include habitat alteration, disturbance, invasive prey species, declining Pacific herring populations and bioaccumulation of trace element contaminants (Carlton et al. 1990, Ohlendorf et al. 1991, Cohen and Carlton 1995, Linville et al. 2002, Merkel et al. 2009).
- Availability of food is hypothesized as a main limiting factor for waterfowl during winter (Lovvorn et al. 2013). Results of carrying capacity models for the San Pablo Bay show that declining profitability as prey becomes patchy and difficult to find may cause scaup and scoters to leave San Pablo before their absolute threshold prey density is reached (Lovvorn et al. 2013).
- Invasive species as food resources may be limiting to diving ducks if they represent lower foraging profitability or an increased risk of contaminant accumulation (e.g., Linville et al. 2002, Wallace, Lee & Luoma 2003), a continual threat given the rate of invasions in the SFB.
- Contaminants are elevated in SFB diving ducks (Takekawa et al. 2002, Eagles-Smith et al. 2009) and may also be limiting as there is evidence that selenium, mercury, and cadmium may influence body condition in scaup and canvasbacks (Takekawa et al 2002). Contaminants did not appear to influence proximate measures of body condition in scoters (De La Cruz 2010), but other studies have tied mercury and selenium to oxidative stress in SFB scoters (Hoffman et al 1998). These subtle effects on condition may later influence survival, migratory timing, and ultimately productivity.
- Disturbance from human recreational and commercial activities is another potential threat that may become exacerbated if available foraging areas for diving waterfowl shrink or move due to climate

related changes. Disturbance can cause waterbirds to expend more energy flying and spend less time feeding, reducing body condition and the ability to migrate and reproduce (Belanger and Bedard 1990, Haramis *et al.* 1986, Bell and Austin 1985). Repeated disturbance may cause waterbirds to shift distribution patterns, forage in less preferred habitats, or emigrate (Schwemmer *et al.* 2011, Havera *et al.* 1992). Responses to human presence can greatly depend on species, bird densities, individual body condition, foraging conditions in the impact area, type of disturbance, and other parameters (Borgmann 2011, Gill 2007, Yasue 2005), and much remains to be learned about how these factors can interact to influence diving duck responses.

- Oil spills are particularly detrimental to diving ducks because these species are most abundant in open waters impacted by spills. For example, scoters and scaup had the highest mortality rates of all birds during the 2007 *Cosco Busan* spill on SFB (Hampton *et al.* 2008, De La Cruz *et al.* 2012). Additionally, many diving ducks species show high winter site fidelity (De La Cruz *et al.* 2009, Zipkin *et al.* 2010) and some species pair on wintering areas; thus, catastrophic events that affect survival rates in SFB could have disproportionate effects on local subpopulations (Esler *et al.* 2000).
- Pacific herring declines may influence body condition and survival of surf scoters and potentially scaup and bufflehead, that rely on roe to build lipid stores during late winter and spring migration (Bayer 1980, Kessel *et al.* 2002, Anderson *et al.* 2009, De La Cruz 2010, Lok *et al.* 2008, 2011, 2012)

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## ENTIRE LIFE CYCLE AND INFLUENCES FROM OUTSIDE THE ESTUARY

Some diving ducks in SFB are boreal nesting species, with greater scaup and surf scoters among the northern-most nesting waterfowl species (Gauthier 1993, Austin *et al.* 1998, Savard *et al.* 1998, Brua 2002, Kessel *et al.* 2002, Mowbray 2002, Takekawa *et al.* 2011), and thus, may be especially susceptible to the effects of climate change. Over the past three decades, the North American Northern Boreal Forest (NBF) has warmed more rapidly than any other region on earth (Serreze *et al.* 2001) and climate-induced changes are predicted to be the greatest in northern ecosystems (Soja *et al.* 2007). Boreal nesting diving ducks have a limited opportunity to reproduce during the ice free period in the Northern Boreal Forest, and changes in invertebrate prey availability (e.g., Visser *et al.* 1998) or in wetland nesting habitats (Smol and Douglas 2007) may be particularly detrimental to the timing of their reproduction (Drever *et al.* 2011). Other threats on the boreal breeding areas are imminent oil, gas, and diamond mining development planned in the NBF (Government of the Northwest Territories; Industry, Tourism, and Investment, <http://www.itl.gov.nt.ca/index.html>). The parkland and prairie pothole regions are also important breeding habitat for lesser scaup, canvasback, ruddy duck, and bufflehead. Clearing and draining of nesting habitat for conversion to agriculture in these regions are of critical concern for diving ducks (Gauthier 1993, Austin 1998, Mowbray 2002).

In addition to wintering in coastal habitats, many diving duck species molt and migrate along coastlines and in estuaries. During these periods they are aggregated in large groups, often in urbanized areas, and may be more vulnerable to threats than when they are dispersed over large, less populated breeding areas. Along coastal migration corridors and molting areas diving ducks contend with factors related to dense human populations including disturbance, contaminant exposure, oil spills, aquaculture, energy development, non-native species invasions, declining Pacific herring stocks (a key prey item for scoters), or shoreline development (Carlton *et al.* 1990, Savard *et al.* 1998, Linville *et al.* 2002, Stick 2005, Bartling 2006, Merkel *et al.* 2009).

Poor winter and spring habitat quality has been hypothesized as one explanation for the observed patterns of continental decline in lesser scaup and other diving ducks. This hypothesis suggests that declines in the quantity and quality of winter and spring habitats have resulted in carry-over effects, such that females are arriving on breeding areas in poorer body condition than historically, resulting in reduced reproductive success and survival (Marra et al. 1998, Anteau and Afton 2004).

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## **FACTORS THAT MAY AFFECT SPECIES RESILIENCE**

Poor winter body condition can affect resilience of diving ducks by making them more susceptible to disease, lowering survival, or exerting cross-seasonal effects on reproduction. Diving ducks typically have high winter site fidelity to particular areas of in SFB (Accurso 1992, De La Cruz 2010, De La Cruz et al. in review). Ensuring important roosting and foraging areas in open bays and baylands are free from disturbance will help diving ducks maintain winter body condition, and thereby improve resilience to other stressors. Subtidal restorations may increase prey abundance for some species, which would lead to improved body condition and increased resilience.

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## **LIKELY CLIMATE CHANGE IMPACTS AND RISKS**

Depending on a given potential climate change effect, impacts and risks will be different for each species. Smaller diving duck species that are more limited by foraging depth may be most influenced by SLR. Prey availability may change for all diving duck species with changes in salinity, fresh water flow, sedimentation, and water clarity. Diving ducks may be forced to disperse to habitat that is not as well suited and prone to more competition, exposure and human disturbance. Changing conditions in the coastal ocean may change distributions of more marine species (surf scoters and greater scaup), as well as the distributions of prey and competitor species in SFB.

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## **MANAGEMENT ACTIONS TO BE CONSIDERED**

- Minimize disturbance to existing roosting and foraging sites, and design recreational trails, boating routes, and restorations to minimize disturbance by people, pets, and predators. Diving duck distributions in SFB may change in relation to SLR and altered prey populations, and changing proximity of important habitat to populated or recreational areas.
- Evaluate the feasibility of using subtidal habitat restorations (e.g., eelgrass, oyster [*Ostrea conchaphila*], other natural substrates) to increase foraging opportunities for diving ducks.
- Manage remaining former salt ponds to maximize their value as roosting and foraging habitat for diving ducks.
- Include large channels and muted tidal areas in restored tidal marsh designs to encourage use by diving duck species.
- Develop habitat models using scenarios of sea level rise, salinity, and sediment supply to help determine areas of SFB that may support diving ducks in the future. This is necessary to direct future management actions for these species.

- Continue long-term survey programs (FWS Midwinter Survey, South Bay Salt Pond Restoration Project Surveys) to evaluate long term trends of diving duck use and numbers in SFB. This type of monitoring is key to an adaptive management approach.
- Participate in the creation of Flyway-wide plans to conserve and protect key winter and migratory stop-over habitat for diving ducks.

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## **UNCERTAINTY AND KNOWLEDGE GAPS**

- More information is needed to understand the role of climatic and oceanic conditions on the variability in use of SFB by different diving duck species and by their fish and invertebrate competitors.
- The effects of disturbance on foraging and habitat use may be variable by species and time of year. Certain species may be more sensitive and certain conditions (low prey availability late in winter) may make disturbance effects more pronounced (Gill 2007).
- More information is needed about connectivity of breeding, migratory, and wintering habitats for diving ducks, in order to understand how each species will respond to changes in the estuary (i.e., some species may redistribute, while others may experience declines in a particular sub-population that winters in SFB).
- Little is known about ruddy duck and bufflehead habitat use in SFB, making their potential responses to climate change more difficult to assess.
- It is uncertain how quickly local sea level rise will occur and thus difficult to determine when habitat changes may occur for diving ducks.
- It is uncertain how sediment deficits in combination with changing climatic conditions will shape the bathymetry of SFB, thus it is unclear which diving duck species will be most influenced by sea level rise.

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# Science Foundation Chapter 5

## Appendix 5.1 – Case Study

### Forster's Tern (*Sterna forsteri*) and California Least Tern (*Sternula antillarum browni*)

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#### DESCRIPTION OF THE SPECIES

This case study considers two tern species that breed within the San Francisco Bay Estuary, Forster's Terns (*Sterna forsteri*) and California Least Terns (*Sternula antillarum browni*). Forster's Terns are medium-sized (140 g) terns that breed in coastal and interior marshes of North America. Forster's Terns can exploit ephemeral habitats, and colony locations often move among years with change in habitat suitability and resource availability. Least Terns are smaller-sized (45 g) terns that breed along beaches and major interior rivers of North America, and winter along marine coastlines in Central and South America. Forster's Terns and California Least Terns breeding in San Francisco Bay tend to use the same nesting colony locations each year.

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#### CRITERIA FOR SELECTION OF THE SPECIES

Forster's Terns were first documented breeding in the estuary in 1948 (Sibley 1952). Since that time, the breeding population size of Forster's Terns has fluctuated annually from 2,400 (Gill 1977) to 5,000 (Ryan 1997) breeding adults, with the current (2005-2014) breeding population estimated at approximately 1,500-3,500 adults (J. T. Ackerman, unpublished). Because monitoring has been sporadic and not systematic, these population size estimates have a large degree of error and likely have limited usefulness for estimating population trends.

Approximately 30% of the Pacific coast population of Forster's Terns breeds within San Francisco Bay (McNicholl et al. 2001, Strong et al. 2004), where the species' breeding ecology has been well studied (Strong et al. 2004, Ackerman and Herzog 2012). Most Forster's Terns nest within former salt evaporation ponds in South San Francisco Bay, particularly the Moffett (ponds A1, A2W, AB1, AB2) and Alviso (ponds A7, A8, A16) Pond Complexes (Ackerman and Herzog 2012). These managed ponds currently provide nesting habitat for over 80% of the terns breeding within the estuary (Strong et al. 2004, Ackerman and Herzog 2012) and are the primary foraging area of adult and juvenile terns (Ackerman et al. 2008, 2009a). Smaller numbers of Forster's Terns nest in managed ponds, on top of duck blinds, and in marshlands of the Napa-Sonoma marshes around San Pablo Bay. Forster's Terns primarily nest on dredge spoil islands within these managed ponds, but other nesting habitats include former dikes that have been

converted into islands (primarily in the Newark Pond Complex), and marshes (such as New Chicago Marsh and Charleston Slough; J. T. Ackerman, unpublished). Forster’s Terns are mainly migratory, breeding in the estuary from April through August and over-wintering further south along the Pacific Coast of California and Mexico (Gill and Mewaldt 1979) with small numbers wintering locally. The first young hatch in late-May and fledge in mid-June (J. T. Ackerman, unpublished).

California Least Terns were first documented breeding in the estuary in 1967 (Chandik and Baldrige 1967 in Gill 1977). California Least Terns are federally endangered and breed at only a few sites in San Francisco Bay. In 2011, a systematic survey of 49 known breeding locations of Least Terns in California estimated 4,826-6,108 breeding pairs statewide. In San Francisco Bay, there were about 300 breeding pairs at Alameda Point, 30 breeding pairs in the Napa-Sonoma Marsh, and 70 breeding pairs at the Hayward Regional Shoreline (Marschalek 2012). Least Terns have also nested sporadically in Eden Landing Ecological Reserve (<10 nests; J. T. Ackerman, unpublished), although no nesting has been documented in this area for the past three years (Marschalek 2012). The largest colony at Alameda Point is highly managed, and is extremely important to the overall statewide population. This site produces a substantial percentage of the fledglings produced in the state (e.g. ~15-22% of the statewide total fledglings in 2009; Marschalek 2010). This site has also maintained a stable number of breeding Least Terns, compared with the large fluctuations in other sites in southern California (Burton and Terrill 2012 and references listed therein) and has been growing at a rate of ~9% per year since 1976 (Elliott et al. 2011).

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## REVIEW OF CLIMATE CHANGE EFFECTS ON THE SPECIES

Forster’s Terns and Least Terns both rely on islands for nesting habitat in San Francisco Bay. Nest survival is considerably greater on islands than in marshes where terrestrial predators can access nests (Ackerman et al. 2014b). Flooding of nesting islands from fluctuating water levels in managed ponds and tidal wetlands is also a significant cause of nest failure for Forster’s Terns.

Forster’s Terns forage predominantly along the bay’s margins within managed ponds and marshes, and to a lesser extent within tidal flats and sloughs (Ackerman et al. 2008, 2009a). Foraging locations for Least Terns from the Alameda colony include marine and estuarine habitats within ~3.5 miles of the colony site (Elliot et al. 2004, Steinbeck et al. 2005). With rising water levels in the estuary associated with climate change, maintaining suitable nesting habitats in close proximity to food resources will be critical to preserving tern nesting colonies. Rising water temperatures and changing habitat conditions associated with climate change also may negatively impact key prey fish species exploited by terns. Northern anchovy (*Engraulis mordax*) may be of particular importance to Least Terns as a small increase in the abundance of anchovy in the diet appeared to lead to an increase in fledging success (Elliott et al. 2007), thus maintaining this resource may be important.

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## OTHER STRESSORS

Studies indicate that Forster’s Terns nesting in San Francisco Bay already face considerable stressors from widespread mercury contamination, and a rapidly expanding predator (California Gulls; *Larus californicus*) population. Although Forster’s Tern nest survival (61%) and egg hatching success (95%) tend to be fairly

high in San Francisco Bay (Ackerman and Herzog 2012), chick survival is low (22%) predominantly due to predation on chicks by California Gulls (Ackerman et al. 2014a). In fact, 54% of all Forster's Tern chick deaths were caused by California Gulls (Ackerman et al. 2014a). Additionally, Forster's Terns and Least Terns both have very high risk to methylmercury contamination. Of 17 species of waterbirds studied in San Francisco Bay, Forster's Terns had, by far, the highest mercury concentrations in eggs, and egg mercury concentrations in Least Terns ranked sixth (J. T. Ackerman, unpublished). Importantly for Forster's Terns in San Francisco Bay, failed-to-hatch eggs and abandoned eggs had higher mercury concentrations than randomly sampled eggs (Ackerman and Eagles-Smith 2008), the likelihood of an embryo being malpositioned increased with egg mercury concentrations (Herring et al. 2010), the probability of an egg successfully hatching decreased with egg mercury concentrations (Eagles-Smith and Ackerman 2010), and the probability of nest survival decreased with egg mercury concentrations (Eagles-Smith and Ackerman 2010). Repeated dredging for the creation of shipping channels adjacent to the Alameda Point Least Tern colony may be exposing the terns to increased risk of contamination by legacy pollutants buried in the sediments (Burton and Terrill 2012 and references listed therein). Warming temperatures, changing water dynamics (precipitation, runoff), and changing habitats associated with climate change have the potential to have a synergistic effect on methylmercury production in San Francisco Bay, thereby possibly increasing the bioavailability of mercury to wildlife, including terns.

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### **LIFE CYCLE CONSIDERATIONS AND POPULATION DYNAMICS**

Little is known about population dynamics of Forster's Terns or Least Terns in San Francisco Bay. The best available data suggest that the Forster's Tern population might be declining (Strong et al. 2004, J. Ackerman, unpublished data) and the small population of Least Terns is likely stable (Elliott et al. 2007). It is unclear what impacts varying climate change scenarios might have on these species, and limited population data currently prohibits robust population projections.

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### **FACTORS THAT MAY AFFECT SPECIES RESILIENCE**

Population resilience will depend on maintaining or enhancing reproductive success, especially nest survival and chick survival. Reducing egg and chick mortality due to predation and contamination, especially mercury, will enhance resilience.

Maintaining island nesting habitat, particularly within managed ponds, will help maintain resilience. Of immediate concern is the conversion of 50% to 90% of the former salt evaporation ponds into tidal marsh habitat under the South Bay Salt Pond Restoration Project. Forster's Terns rely on the project's managed pond habitat for foraging and nesting, and reducing its availability will likely result in a smaller tern population. This large-scale wetland restoration project will likely have a larger influence on the bay's Forster's Tern population than any changes related to climate change. Maintaining managed ponds and associated nesting islands preferred by Forster's Terns will be important (particularly project ponds A1, A2W, AB1, AB2, A7, A8, and A16), as will the creation of new nesting islands suitable to terns.

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### **LIKELY CLIMATE CHANGE IMPACTS AND RISKS**

Forster's Terns and Least Terns rely on wetland habitats along the margins of San Francisco Bay. In particular, managed ponds and their associated dredge spoil islands are critically important for tern foraging

and nesting. Management to enhance reproductive success is likely the most critical factor for maintaining or increasing populations in San Francisco Bay, and loss of island nesting habitat and methylmercury contamination both have been shown to impair reproduction. Because most Forster's Terns nest in South San Francisco Bay, reduction in managed pond habitat (former salt evaporation pond) by as much as 90% by the South Bay Salt Pond Restoration Project will likely have a dramatic influence on the viability of tern populations.

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### MANAGEMENT ACTIONS TO BE CONSIDERED

Management should focus on enhancing reproductive success through improved nest survival and chick growth and survival. Therefore reducing nest predation rates and methylmercury bioaccumulation is recommended. Reducing nest predation rates could be accomplished by increasing the terns' island nesting habitats in close proximity to food resources, but away from colonies of the growing population of California Gulls (Ackerman et al. 2006, 2009b, 2010a, Ackerman et al. 2014a). Reducing methylmercury contamination is more challenging, but enacting management actions in wetlands to reduce methylmercury production and bioaccumulation may be possible (Ackerman et al. 2010b, Ackerman et al. 2012).

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### UNCERTAINTY AND ITS SOURCES

The demographic response of Forster's Terns and Least Terns to climate change is not known, especially with regard to survival rates. One thing that is more certain, however, is that enhancing or increasing island nesting habitat within wetlands along the bay's margins should benefit breeding terns and allow colonies to move to new nesting locations as resources fluctuate spatially and temporally.

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### IMPORTANT DATA GAPS/NEEDS

The main data gap is uncertainty regarding what impacts the South Bay Salt Pond Restoration Project may have on the breeding population of Forster's Terns in San Francisco Bay. Of particular importance is understanding where and how to construct nesting islands that will be used by terns and promote sustainable reproductive success rates. Additionally, more information is needed about the benefits of not converting (into tidal marsh habitat) the most productive current tern nesting locations. Demographic responses of Forster's Terns and Least Terns to climate change is not known, especially with regard to nest and chick survival rates, and deserves more detailed study.

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# Science Foundation Chapter 5

## Appendix 5.1 – Case Study

### Harbor Seal (*Phoca vitulina*)

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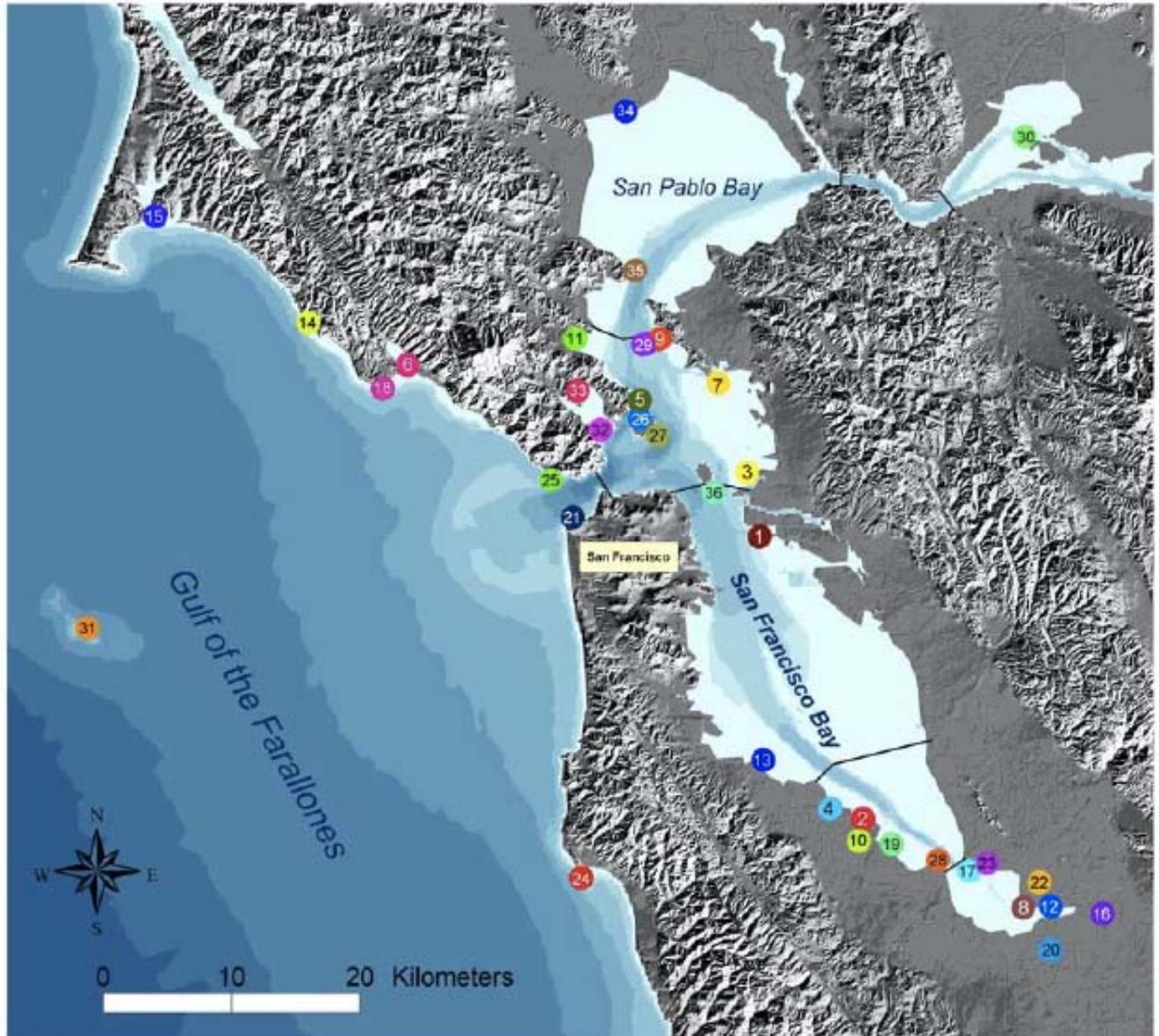
#### DESCRIPTION OF THE SPECIES

Harbor seals (*Phoca vitulina*) are the only marine mammal present in San Francisco Bay year round where they rest ashore on islands, tidal rocks, mudflats, and sand bars. Other marine mammals may visit the Bay seasonally to forage (California sea lions, *Zalophus californianus*) or occur at the mouth of the bay while migrating or foraging (gray whales, *Eschrichtius robustus*; harbor porpoise, *Phocoena phocoena*; bottlenose dolphins (*Tursiops truncatus*), but harbor seals are year round residents. They use terrestrial areas (called haulouts) daily to rest between foraging trips and annually during breeding season (March-June) and molt (June-July). While substrate may vary, haulouts are generally characterized by gentle slopes, easy access to deep water and proximity to food (Allen 1991). The seals use the same haulouts year round and year after year, and generally seals have been using the same sites in the Bay for decades and in some cases for more than 60 years. Individual seals may frequent multiple haulouts within the Bay, and also move outside of the bay to coastal sites to the north and south. In the past, seals hauled out on a greater number of sites within the Bay, but researchers have concluded that seals abandoned some of these, likely because of human development and disturbance (Bartholomew 1949, Grigg et al. 2002).

During the breeding season, seals are more abundant onshore while nursing pups. Harbor seal pups are unusual for a pinniped because they are precocious and can swim at birth. Mating occurs in the water and male seals have “maritories” which are underwater territories near female haulouts that they defend from other males. Although their vocalizations are not elaborate, underwater vocalizations are apparently important to seal breeding (Hayes 2004a, Hayes 2004b).

Within a day, seals are generally more abundant onshore during the day time than at night, but may be present at haulout sites any time of the day or night when tide levels expose the substrate (rocks or tidal sandbars). Seals rest onshore for an average of 7 hours per day, and when seals are with pups or molting, average time onshore increases to 10-12 hours per day depending on availability of haulout space (Allen 1988).

There are nearly 30 haulout sites within San Francisco Bay, but the number of seals at most of the sites is small. There are 3 major sites, with more than 100 seals, located in the north, central and south bays (see Grigg et al. 2004). The number of seals in the Bay varies by site and by season. More seals are present overall in the Bay during the winter foraging period than during the spring breeding season. This pattern in the Bay is different from that reported at remote, nearby coastal sites where a higher abundance of seals occurs during the breeding and molt seasons (Codde and Allen 2013). Large concentrations of spawning Pacific herring (*Clupea pallasii*) and salmonids likely attract seals into the Bay during the winter months.



Map of current and historical haulout locations in San Francisco Bay and along the adjacent coastline from Green et al (2006).

1. Alameda Breakwater; 2. Bair Island; 3. Bay Bridge Sandspit; 4. Belmont Slough; 5. Bluff Point, Tiburon; 6. Bolinas Lagoon; 7. Brook's Island; 8. Calaveras Point; 9. Castro Rocks; 10. Corkscrew Slough; 11. Corte Madera; 12. Coyote Creek; 13. Coyote Point; 14. Double Point (PRNS); 15. Drakes Estero (PRNS); 16. Drawbridge; 17. Dumbarton Point; 18. Duxbury Reef; 19. Greco Island; 20. Guadalupe Slough; 21. Land's End; 22. Mowry Slough; 23. Newark Slough; 24. Pillar Point; 25. Pt Bonita; 26. Pt. Ione, Angel Island; 27. Pt Blunt, Angel Island; 28. Ravenswood Point; 29. Red Rock; 30. Ryer Island (Suisun Bay); 31. SE Farallon Island; 32. Sausalito Boatworks; 33. Strawberry Spit; 34. Tubb's Island; 35. Two Sisters; 36. Yerba Buena Island.

Harbor seals forage within and outside of the Bay and individual seals have been documented foraging in the Sacramento River as far north as Sacramento. Most seals, though, forage within the three primary bays with the largest haulout sites (Grigg et al. 2012). They are opportunistic eaters whose diet varies seasonally and even regionally within the bay (Gibble and Harvey 2015, Harvey and Torok 1994). Fish are the primary prey item but seals also forage on invertebrates such as shrimp (*Crangon* spp) and market squid (*Loligo opalesens*). More detailed information about harbor seal food habits and historical distribution within the bay can be found in the Goals Project 2000.

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## CRITERIA FOR SELECTION OF THE SPECIES

Harbor seals have been studied extensively over the past 50 years in SF Bay (Bartholomew 1949, Grigg et al. 2004). Globally, they are vulnerable to perturbations such as human disturbance, prey shifts, pollutants and disease outbreaks, and these same perturbations have affected seals in SF Bay (Paulbitski 1975, Riseborough et al. 1980, Harvey and Torok 1994, Allen 1991, Greig et al. 2011, Grigg et al. 2012).

Harbor seals are native to San Francisco Bay and are protected under the Marine Mammal Protection Act (MMPA) which was passed in 1972. In the 1970s and 1980s after passage of the MMPA, the harbor seal population in CA increased along the central CA coast with the growth slowing in the 1990s (Sydeman and Allen 1999). Within the bay, the population has been relatively stable or slowly increasing depending exactly where and when you count (Grigg et al. 2004). The current population estimate for the state of CA, based on a 2009 aerial count, is 30,196 seals (95% CI = 22,745–37,647; Harvey and Goley 2011) and within the bay, numbers are estimated at 500 – 600 seals (Grigg 2004, Lowry 2008). Seal numbers are monitored during the breeding and molt seasons in the Bay and along the coast by a network of biologists, and the data are summarized annually in a report (see website: [http://www.sfnp.org/harbor\\_Seals/](http://www.sfnp.org/harbor_Seals/)).

Harbor seals are of conservation concern and considered vulnerable to climate change due to 1) anticipated impacts on their habitat (intertidal zone) and lack of alternative habitats, and 2) indirect effects on food availability, haul out space, increased exposure to diseases and increased interactions with humans.

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## OTHER INFORMATION ABOUT THE SPECIES

A high number of SF Bay seals (~40%) exhibit a red coloration compared to other locations along the CA coast or even globally (Allen et al. 1993). This coloration is a result of iron oxide precipitation on the hair. Each summer when the harbor seals molt, the red hair is shed and replaced with silver hair; however, within a couple of weeks the rusty color returns again. While red seals often have brittle fur and short whiskers, there is no evidence of ill health (McHuron 2012) or differences in heavy metal accumulation in muscle or liver (Moser 1996). Nevertheless, one study suggested there was a correlation between red seals and pollutant loads (Kopec and Harvey 1995).

Harbor seals are easily disturbed by human activities adjacent to their haulout sites. Harbor seals are able to adapt to human presence which has enabled them to continue to breed in highly industrialized areas such as the San Francisco Bay; however, there is a threshold at which seals no longer tolerate human activities and will alter their haul out patterns or abandon a site if disturbances are chronic or if habitat is altered. In one study, seals hauled out more at night than during the day at one site in the Bay, likely because of disturbance (Grigg et al. 2002). Abandonment of haulout sites has also been documented within the Bay (Allen 1991). Alternatively, seals have also been documented using new haulout sites within the Bay where there are fewer disturbances (Lowry et al. 2008).

Pollutants in the Bay have been a concern since the 1970's when aborted fetuses were reported at multiple locations within the bay (Risebrough et al. 1980) and organochlorine concentrations have been detected in harbor seal tissues at levels known to cause reproductive and health effects in European harbor seals (Kopeck and Harvey 1995, Neale et al. 2005). Mercury contamination is also an ongoing concern for harbor seals in the bay area (Brookens et al. 2007, Brookens et al. 2008, McHuron et al. 2014).

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## **REVIEW OF CLIMATE CHANGE EFFECTS ON THE SPECIES**

### 1. Loss of habitat with sea level rise or during extreme storm events.

Sea level rise, large storm events and erosion along the shoreline are predicted to increase and alter shorelines within the Bay. Loss of habitat is predicted to have a large effect on the intertidal haulout habitat of harbor seals (Largier et al. 2010). These sites are used for breeding and resting. The existing numbers of haulout sites where seals give birth are limited and there are few alternative sites available now or likely under the various sea level rise scenarios. Loss of this habitat will severely reduce the number of seals breeding in the bay, although seals may travel from elsewhere into the bay to forage.

### 2. Prey shifts with warming sea water temperature.

Pinniped populations are sensitive to environmental perturbations such as El Niño events, which cause shifts in their prey. During ENSO events, harbor seals produce fewer pups (Allen et al. 1989, Sydeman and Allen 1999, Becker et al. 2011). Since seals prey on what is locally abundant, changes in prey availability due to ENSO events likely reduce food for seals, possibly resulting in their inability to maintain a pregnancy. Intense ENSOs such as in 1998, may affect foraging and survival of adults as was documented in elephant seals (LeBoeuf et al. 2000). Reductions in prey availability could decrease body condition which could affect reproduction (decreased pregnancy rate) and health (decreased immunity, increased susceptibility to disease).

### 3. Pathogen range shifts with warming sea water temperature.

Harbor seals are susceptible to infection from a number of disease causing pathogens. As waters warm, many of these pathogens have the opportunity to expand into new geographic areas. As an example, morbilliviruses such as phocine distemper (PDV) have had devastating consequences on harbor seals in Europe (Härkönen et al 2006). To date, there have been no morbillivirus outbreaks in the Pacific and PDV has not been detected in SF Bay harbor seals (Greig et al. 2014). Recently, however, the strain of PDV that caused >30,000 deaths in harbor seals in Europe was detected in sea otters in Alaska (Goldstein et al. 2009). It is hypothesized that decreasing ice conditions in the arctic may have altered seal haulout and migration patterns and allowed contact between Atlantic, Arctic, and Pacific sub-species of harbor seals and other pinniped species that had not occurred previously. This highly infectious virus has the potential to infect the immunologically naïve harbor seals of the Pacific.

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## **OTHER STRESSORS**

### 1. Disturbance (hikers, kayakers, clammers, cars, motor boats, aircraft) and other interactions with humans (pollution, boat strike).

Even when disturbance does not result in direct mortality or haulout site abandonment it can have more subtle effects. For example, more pups are born at sites with lower disturbance rates (Codde et al. 2013). First year mortality among harbor seal pups also appears to be reduced in San Francisco Bay compared with nearby locations although the reasons for that are not clear (Greig 2011).

2. Disease (algal toxins, bacteria, viruses, protozoal, pollution).

Pollution has been a concern of harbor seals for decades because of the elevated levels of PCBs and DDEs in the seals of the Bay (Risebrough et al. 1980, Kopec and Harvey 1995). Lower reproduction and aborted fetuses were speculated to be associated with these elevated pollutant loads. More recently, pollutant concentrations in harbor seal tissue have been associated with immune systems effects, congenital defects, and first year survival (Neale et al. 2002, Neale et al. 2005, Harris et al. 2011, Greig 2011).

Combinations of stressors have been implicated in major population declines of harbor seals in other locations. Declines in harbor seals in Scotland since 2000 have been attributed to a combination of factors including human caused trauma, biotoxins, and possibly competition for food with other seal species (Thompson et al. 2001, Bexton et al. 2012, Hall and Frame 2010). During the 1988 PDV outbreak in the North Sea, contaminant concentrations in the blubber of seals that died from PDV were greater than in seals that survived the outbreak (Hall 1992). Synergistic effects are also possible among these additional stressors; embryonic exposure of zebrafish (*Danio rerio*) to DDT enhanced their susceptibility to seizures induced by domoic acid suggesting that in utero contaminant exposure could have the same effect on marine mammals (Tiedeken and Ramsdell 2009).

3. Human land use will also continue to affect the bay in terms of chemical pollutants, and increased nutrients entering the bay. For example, as use of new flame retardant chemicals increases, these emerging chemical may enter the Bay food chain (Klosterhaus et al. 2012).

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## LIFE CYCLE CONSIDERATIONS

One pup is born per year per female (March – May), females are reproductive at 3 to 5 years of age. Females are presumed to pup every year thereafter with no data on senescence. Females can live to 15 or 20 years old. After birthing, females nurse pups for 30 days and then wean them. After weaning, pups forage alone, and first year survival is highly variable depending on year and location (Lander et al. 2002, Oates 2005, Greig 2011). Females skip pupping during ENSO years. Males do not contribute to rearing pups. Mortality can be as high as 80% during the first year of life making this is a critical life stage for the species.

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## FACTORS THAT MAY AFFECT SPECIES RESILIENCE

Harbor seals are very resilient to the occasional bad year (El Niño for example), but not to sustained, chronic change/disturbance which can result in abandonment of habitat areas (Bartholomew 1949, others). If alternate suitable habitat is available near disturbed haul out sites, seals may shift to new haul out sites (Allen et al. 1985).

Harbor seals within the bay have exhibited resilience to sustained predictable disturbance by switching from a diurnal to nocturnal haulout pattern as a result of activity on the Richmond Bridge (Grigg et al. 2002). Seasonal closure of haulout sites has resulted in an increase in seal numbers and pupping (for example, Point Bonita in SF Bay, Codde et al. 2013).

Harbor seals also have exhibited an ability to switch prey within the Bay, eating non-native yellow gobies when they became more abundant (Gibble and Harvey 2015, Harvey and Torok 1994).

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## **LIKELY CLIMATE CHANGE IMPACTS AND RISKS**

1. Loss of resting and nursery habitat with sea level rise or during extreme storm events
2. Prey shifts with warming sea water temperature, ocean acidification
3. Changes in disease ecology/risk – pathogens may undergo range shifts with warming water and changes in salinity.

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## **MANAGEMENT ACTIONS TO BE CONSIDERED**

Short-term v. long-term recommendations – focus on thresholds

1. Maintain current mudflat/rocky habitat and buffer from human population (ie. maintain or decrease level of disturbance).

North Bay: Castro Rocks, Ryer Island, Corte Madera mudflats, East Brother Island

Central Bay: Yerba Buena Island, Angel Island

South Bay: Mowry/Newark/Alviso Sloughs; Bair Island/Corkscrew Slough

This includes restoration or other environmental management efforts which have the possibility to reduce disturbance of seals. If sea level rises, this will result in the inundation and loss of current habitat, but more will form if human development is maintained far enough away from the shoreline. For example, the restoration of south bay salt ponds to wetland habitat is a great opportunity for creating seal habitat including potential haul out sites for harbor seals. They currently haul out in several sloughs on the edge of the old salt ponds. Additionally, there is potential alternative shoreline habitat on Red Rock Island (private land) and Angel Island (California State Park) if disturbances are reduced.

2. Monitor seals within the bay for changes in behavior, numbers, and habitat use and develop a seal monitoring program in the bay to address some of the data gaps highlighted below.

For seal populations along the coast, the National Park Service recommended the following (Allen 2011):

1. Monitor the population and links to environmental conditions
2. Reduce disturbance and fisheries interactions
3. Restore pinniped habitats
4. Communicate with the public

## UNCERTAINTY AND KNOWLEDGE GAPS

There is uncertainty about the harbor seal response to climate change stemming in part from the data gaps in knowledge of harbor seal population dynamics within SF Bay (detailed below), and also from the unpredictability of these animals. Additionally there is uncertainty regarding the effects of climate change on prey availability and disease ecology. This uncertainty does not change the importance of maintaining haulout areas to increase seal flexibility and resilience.

### Important Data Gaps

We are missing the data needed to monitor and model the population of seals within San Francisco Bay:

1. Basic information on number of seals and habitat use within the bay. Are there potential new haulout locations within the bay where numbers are increasing already (for example, Ryer's Island in the north bay), or locations that have the potential to absorb seals (Red Rock Island and Angel Island)?
2. Demographic data - Pregnancy rate, mortality and survival rates
3. Movements and habitat use
4. Synergistic effects of disease, pollutants, and disturbance on health and survival
5. Diet. Although there have been periodic studies on the diet of seals in the bay, there have been no long term studies to document seasonal and inter-annual changes in diet. For example, how does diet change during ENSO events?

All of these gaps would be optimally addressed by a comprehensive, ongoing monitoring program.

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# Science Foundation Chapter 5

## Appendix 5.1 – Case Study

### Longjaw mudsucker (*Gillichthys mirabilis*)

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#### DESCRIPTION OF THE SPECIES

The longjaw mudsucker (*Gillichthys mirabilis*) is a resident estuarine fish, ranging from Mexico to Humboldt Bay, California, USA, and is one of the most abundant fishes in high intertidal salt-marsh habitat (Desmond et al., 2000; Talley 2000; West and Zedler 2000). The Longjaw mudsucker depends on high intertidal complex dendritic creeks within salt marsh primarily “pickleweed” [*Salicornia virginica* now taxonomically known as (*Sarcocornia pacifica*)]. The fish reside within burrows in soft sediments and is the only fish species that can remain in intertidal creeks during low tide when the creeks completely de-water. The mudsucker can tolerate life out of water by having vascularized buccal cavities for uptaking oxygen from the air. Mudsuckers have a wide environmental tolerance, and are able to tolerate freshwater and salinities as high as 90-ppt for periods of a few days to a week, and temperatures from 9-35 C° (Lonzarich and Smith 1997, Moyle 2002). Longjaw mudsuckers are benthic consumers, most commonly eating bottom-dwelling invertebrates, such as amphipods, isopods, and small fish. Males will guard burrows and display their long maxillae, hence their common name, to attract females. Spawning occurs predominantly from late winter to early spring, with pelagic larvae settling to the benthos approximately two months after hatching. Juveniles (<80mm) spread out into many different habitats during summer, while adults tend to spend most of their lives in a single creek habitat, not straying more than a few meters from their burrows (Hobbs unpublished data). With such a high degree of site fidelity, the longjaw mudsucker completes its life cycle in a single marsh making it an excellent candidate as a sentinel species of saltmarsh habitat quality (Yoklavich et al., 1992).

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#### CRITERIA FOR SELECTION OF THE SPECIES

Since the longjaw mudsucker depends on sinuous intertidal creeks in pickleweed marsh habitats, it may be especially vulnerable to sea level rise and thus a good candidate species when considering effects of climate change on tidal marsh habitats of San Francisco Bay. Like the other species depending on pickleweed marsh habitats such as the salt marsh harvest mouse and clapper rail, the species is particularly vulnerable to sea level rise because most remnant salt marsh habitats in South San Francisco Bay are backed by large levees, and thus limited in ability to adapt or move upland as sea level rises. The species could be considered a sentinel of salt marsh habitat health as it is the only fish species that occurs in intertidal creeks, taking refuge in burrows or in soft sediments during low tide when the creeks are dewatered. The longjaw mudsucker is rarely found outside of intertidal pickleweed marsh, primarily only after larvae settle from the

water column and fish search out available creek habitats (Hobbs et al 2012). While the species does occur outside of San Francisco Bay, it is of conservation concern, as in many estuaries the salt marsh habitats have been highly altered, contaminated and invaded by non-native species. Moreover because the larvae appear to remain in upstream slough habitats, and are sensitive to high salinities, it is unlikely that connectivity among adjacent estuaries along the California coast is sufficient to sustain populations that are lost, and genetic distinctness is likely to exist among different estuaries.

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## OTHER INFORMATION ABOUT THE SPECIES

Because the species utilizes intertidal creeks for the majority of their life are often found within a few meters of original capture in subsequent recapture events, and males have been observed to guard burrows, the population dynamics can be characterized to be highly density dependent. This was apparent in mark-recapture studies conducted in south bay among creek habitats of different length. While catch per trap would often be much higher in the shorter creeks, abundance estimates from the mark-recaptures studies revealed similar or greater abundance in the longer creeks as in the short creeks, individuals were much more likely to be recaptured during subsequent surveys (Hobbs et al. 2012). The creation of new marsh habitat could result in an increase in total population abundance. This was evident in the south bay salt pond restorations, as ponds were restored to tidal inundation, new recruits could be found in newly opened pond habitats such as A6 and in ponds that had been open for several years and begun to return to pickleweed marsh habitats like pond A21, similar numbers of adults and juveniles could be found inside the pond relative to adjacent sites (Hobbs et al. 2012).

The species is also a very important component of the food web as they are a sought after prey item for many predatory species such as leopard sharks, bat rays and striped bass. Body morphology and lipid content would support this supposition as they are soft bodied, without any spinous rays for defense and contain high body lipid content relative to similar prey species such as the yellowfin goby and staghorn sculpin (Hobbs *unpublished data*). Longjaw mudsucker as prey would form a very important linkage between the intertidal marsh habitats and adjacent slough habitats, as they feed exclusively in marsh habitats and are preyed upon by benthic and pelagic slough oriented predators. Indeed the species is a favorite among bait fisherman in San Francisco Bay.

The status of the population is currently unknown; however considering their obligate dependence on pickleweed marshes, and the significant loss of this habitat in the last 100 years the population is surely a remnant of what it once was in San Francisco Bay. While pickleweed marsh extent is thought to be 10% of its pre-industrial range (Atwater et al. 1979), not all marshes have abundant longjaw mudsucker populations. In 2006, (Hobbs *unpublished data*) conducted a bay-wide survey including the Napa-Sonoma Marsh in the north bay, China Camp (Muzzi) Marsh, Stege Marsh in central bay and the Newark Slough marsh, the largest remaining pickleweed marsh in San Francisco Bay. They found the Napa-Sonoma Marsh to be devoid of longjaw mudsucker and only found the invasive yellowfin goby, the China Camp population to be small but present, the Stege Marsh population to have disappeared (McGourty et al 2009), and only the Newark marsh to have large numbers of longjaw mudsuckers. From 2010-2012 several sites within the Alviso Marsh complex, Ravenswood Marsh, Bair Island outer marsh and Eden Landings Whales Tale Marsh, were monitored monthly. Large numbers of fish were found in the Alviso complex, fewer in the Ravenswood Marsh and very few in the Whales Tale Marsh, but only a few individual longjaw mudsuckers were found in the Bair Island outer marsh over the two year study (Hobbs et al 2012). These surveys suggest that the population status is likely to be poor overall with only a few marshes harboring significant, but fragmented populations.

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## REVIEW OF CLIMATE CHANGE EFFECTS ON THE SPECIES

Current sea level rise predictions for San Francisco Bay by 2030 given the current observed rate of sea level rise (Stralberg et al. 2011), places the pickleweed marshes in jeopardy due the dual effects of increased inundation and salinity (Woo and Takekawa 2012; Schile et al. 2011). By 2050 many of the marshes may succumb to inundation due to the leveeing of most upland habitats surrounding these marsh habitats. Some salt marsh habitats may keep pace with sea level rise through sediment and organic material accretion, however this could lead to further fragmentation of populations and leave them even more vulnerable to other population drivers (Stralberg et al. 2011).

It is less clear how sea level rise and variable salinity and freshwater outflow will influence recruitment of the species. Recruitment appears to be greater during years of higher freshwater outflow (McGourty et al. 2009, Hobbs et al. 2012, however future climate may have increased variability of freshwater outflow and earlier timing of freshwater outflow which would have unpredictable effects on reproductive timing. They spend other than that month or two in the plankton and sloughs trying to find a burrow. The future climate may also have higher air temperatures leading to higher shallow water temperatures which may have a significant effect on shallow intertidal habitats where the longjaw mudsucker lives. While the species is tolerant of extreme temperatures for short periods of time, longer term changes to the intertidal habitats due to increased air temperatures could lead long term thermal stress to the fish, which could have negative effects on the long term condition of the fish (Somero 2010). However a concomitant rise in sea level could ameliorate intertidal temperature effects with flooding of cooler water, but again the interplay of these two effects is unpredictable at this moment. Moreover, it is predicted that along the California coast ocean water temperatures may be cooler due to increased ocean upwelling (Snyder et al 2003).

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## OTHER STRESSORS

The longjaw mudsucker, being a species with a wide physical environmental tolerance, suggests that habitat availability is the driver of the species population dynamics. Indeed the longjaw mudsucker being the only fish obligate to pickleweed marsh habitats, have likely undergone significant long-term population declines in the San Francisco Bay-Estuary as most of the pickleweed marsh habitat was dyked and utilized for salt production in the early 1900's. (Atwater 1979) The population of longjaw mudsucker may also be jeopardized by contaminants in salt marsh habitats. In the San Francisco Bay-Estuary populations of longjaw mudsucker have been shown to be much lower in abundance in highly contaminated marshes relative to less contaminated marshes (McGourty et al. 2009). The species may also be susceptible to disease induced by a parasitic microsporidian protozoan (*Kabatana newberryi*) that was first found in the tide water goby and has also been observed in high prevalence in populations of longjaw mudsucker in San Francisco Bay (McGourty et al. 2007. Hobbs *personal observations*). The species also appears to compete for limited habitat space with or be preyed upon by the invasive species the yellowfin goby (*Acanthogobius flavimanus*) (Brooks dissertation 1999, Hobbs *unpublished data*).

## **FACTORS THAT MAY AFFECT SPECIES RESILIENCE**

Given the species wide range of environmental tolerance, the persistence of the species in San Francisco Bay is dependent upon the existence of pickleweed marsh habitats, which in at least some place in the San Francisco Bay may be able to accrete enough sediment and organic material to keep pace with sea level rise.

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## **MANAGEMENT ACTIONS TO BE CONSIDERED**

The South Bay Salt Pond Restoration Program is currently restoring former salt pond habitats to tidal marshes, which could have major ameliorating effects on many of the potential negative impacts of climate change on the longjaw mudsucker population in San Francisco Bay as well as many other marsh dependent species. The project goal is to restore over 15,000 acres of salt ponds to tidal action which could have a significant impact of the rate of sea level rise for South San Francisco Bay and increase salt marsh habitat significantly (South Bay Salt Pond Restoration Program Adaptive Management Plan 2007). However the project is in dire need of long-term funding to meet the restoration goals, and funding of the project is highly recommended. Also acquiring more salt pond habitat from Cargill to restore to tidal marsh.

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# Science Foundation Chapter 5

## Appendix 5.1 – Case Study

### Marsh Macroinvertebrates

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#### DESCRIPTION OF THE SPECIES

This case study includes representative macroinvertebrates that live in the marsh plain, its associated channels and pannes (ponds), and the marsh-upland transition zone. While less visible than animals such as birds, invertebrates play important roles in physical and biological processes (e.g., burrowing activity and channel bank erosion, and detritivores breaking down organic matter) and are important food resources for higher trophic animals. Common invertebrates in these habitats include plant-hopper (*Prokelisia marginata*), beach hopper (*Traskorchestia traskiana*), pygmy blue butterfly (*Brephidium exilis*), inchworm moth (*Perizoma custodiata*), western tanarthus beetle (*Tanarthus occidentalis*), salt marsh mosquitoes (*Aedes* spp.; Maffei 2000a, Maffei 2000b, Maffei 2000c), crabs (native *Hemigrapsus oregonensis* and introduced *Carcinus maenas*), copepods, snails (e.g. native California horn snail *Cerithidea californica* and introduced *Ilyanassa obsoleta*, *Myosotella myosotis*), polychaetes (e.g. *Capitella* spp., *Eteone californica*, *Neanthes brandti*), small clams (*Macoma petalum*/*M. balthica*), and corophiid amphipods (Cohen 2011, Race 1982, Robinson et al. 2011). Some common species were described in detail in the San Francisco Bay Goals Project *Species and Community Profiles* (Goals Project 2000).

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#### CRITERIA FOR SELECTION OF THE GUILD

This broad group of organisms serves a variety of ecological roles. Some species are food for songbirds, shorebirds, waterfowl, fish, or small vertebrates (Goals Project 2000, Dean 2005, Robinson et al. 2011). Others pollinate plants or scavenge dead vegetation. The introduced isopod *Sphaeroma quoianum* can increase erosion of marsh channel banks through its burrowing (Galley and Levin 1999). Some species distributions are restricted to narrow habitats. Others provide a link between the marsh and upland habitats by traveling between them (Maffei 2000d). Many are poorly studied; therefore, this case study focuses on species for which we could find sufficient information. Climate change may create negative impacts on some species and positive impacts for others, depending on their ecology and the physical changes experienced by marshes in coming decades.

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#### OTHER INFORMATION ABOUT THE SPECIES

Each invertebrate species tends to have a preference for particular environments within the marsh, such as the marsh plain, channels, or pannes. For example, at China Camp Marsh in the North Bay, corophiid

amphipods and the bivalve *Macoma petalum* were found mostly in large channels, while the non-native mussel *Geukensia demissa* inhabited the lower sections of channels at the edge of the marsh (Robinson et al. 2011). While the marsh contains many species, a few of them tend to make up the majority of the numbers and biomass, and in San Francisco Bay marshes many of the most abundant species were introduced from other parts of the world.

Many factors influence where a particular species will be found in a marsh, including physical structure, predation, competition, larval settlement and survival, and disturbance (Kneib 1984). Race (1982) studied competition between California horn snails and introduced *Ilyanassa obsoleta* snails in San Francisco Bay. She found that the marsh pannes (small ponds) provide a refuge for California horn snails when they are outcompeted by *Ilyanassa* on mudflats. Life cycles also influence invertebrate distribution. Many species are dormant during the winter. Species that have larval stages may use different parts of the marsh at each stage, for example with an aquatic larval stage and adults living on vegetation.

Invertebrates have developed a range of physiological and behavioral adaptations for life in the marsh. These include adaptations to survive fluctuations in salinity, temperature, and water levels on a daily to seasonal basis (Parker et al. 2012).

Tidal marshes are highly productive ecosystems and marsh-derived organic matter and nutrients are important food subsidies for aquatic and coastal ecosystems (Teal 1962; Odum 1980). Relatively few experimental studies have been conducted on tidal marsh food webs and trophic structure in San Francisco Bay, though see Howe and Simenstad (2007). Stable isotope analysis is a common tool that uses naturally occurring elements to trace food web sources and pathways. Stable isotope analyses showed that food webs for marsh invertebrates (such as filter feeding mussels) depended mostly on inputs from the marsh itself rather than on nutrients from the bay, even in newly restored marshes (Howe and Simenstad 2007, 2011), suggesting that inter-marsh connectivity can play an important role in subsidizing food webs in early marsh restorations. Juvenile fishes forage on insects that fall into the water column from tidal marsh plant canopies, as well as tidal marsh-derived materials that end up in aquatic food webs (Howe and Simenstad 2011; Herbold et al. 2014).

Grenier (2004) used stable isotope analyses within a tidal marsh in San Francisco Bay and conclude that marsh plain macroinvertebrates have fairly simple food chains extending from aphids to wolf spiders (Grenier 2004). The diets of song sparrow (*Melospiza melodia*), California black rails (*Laterallus jamaicensis coturniculus*), and California voles (*Microtus californicus sanpabloensis*) appeared similar, based mostly on the aerial pathway (insects) rather than the benthic (ground) food web pathway (Grenier 2004). This food chain is spatially separated and distinct from benthic invertebrates in the mudflats, which are generally supported by benthic microalgae (Neira et al. 2005).

Food web studies have confirmed the role of cordgrass (*Spartina spp.*) detritus in the diet of surface and subsurface detrital consumers (Levin et al. 2006). Leaf-hoppers, which feed on plant sap, also had a stable isotope composition similar to *Spartina* cordgrass, indicating that cordgrass is their primary food resource. Several recent studies examined the effects of invasive hybrid *Spartina* (a hybrid between San Francisco Bay native *Spartina foliosa* and U.S. east coast species *Spartina alterniflora*) on invertebrate food webs. Hybrid *Spartina* created numerous physical and biological changes to invaded marshes that reduced species richness and density and further changed macroinvertebrate community structure from surface microalgae feeders (i.e., crustaceans and bivalves) to one dominated by belowground plant detritus consumers (oligochaetes; Levin et al. 2006, Neira et al. 2007). The reduction in bivalves, amphipods, and polychaetes within invasive hybrid *Spartina* zones is also a concern because these invertebrates are important food for migratory

shorebirds (Levin et al. 2006, Neira et al. 2007). Furthermore, while hybrid *Spartina* added more plant detritus into the marshes where it invaded, invertebrates such as crabs, clams, and mussels did not ingest it, indicating that invasive hybrid *Spartina* did not create a new food source (Brusati and Grosholz 2009) for the key invertebrate prey resources for migratory birds.

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## REVIEW OF CLIMATE CHANGE EFFECTS ON THE SPECIES

These species already live with periodic natural stresses in the form of tidal cycles and changes in salinity over a day or season. Therefore, the scale of climate change impacts may depend on whether changes are outside the range of variation these species already experience and their ability to adapt. In addition, some stresses caused by climate change may be long-term, such as loss of high marsh, while others may be more periodic, such as changes in inundation periods or seasonal salinity. The impacts described below for different climate change scenarios are educated guesses extrapolated from the available information on macroinvertebrate tolerances to temperature, salinity, and other environmental factors. The description below focuses on the scenarios that are most likely to have an impact on macroinvertebrate communities.

All scenarios: The effect of changes in temperature extremes and temperature fluctuations may depend on how the current climate of San Francisco Bay marshes compares to the physiological tolerances of particular species (Deutsch 2008, Kingsolver et al. 2011). In general, increased temperatures in water, sediment, or ambient air could each affect invertebrates, as the habitats of these organisms differ at different life stages or different parts of the tidal cycle. Not all changes may be negative; warmer temperatures could benefit insects in temperate latitudes by increasing population growth rates (Deutsch 2008). However, if marsh plants that are hosts for insect larvae shift their bloom periods, this could create a disconnection between when eggs are laid and when resources are available for larvae, reducing survival rates (Durant et al. 2007).

Two of the larger macroinvertebrates in salt marshes are native shore crabs (*Hemigrapsus oregonensis*) and invasive European green crabs (*Carcinus maenas*). Both species can tolerate a wide range of temperatures. Shore crabs live in estuaries from Alaska to Baja California, putting San Francisco Bay in the middle of their geographic range and temperature conditions. Therefore, it is likely that they will be able to tolerate increasing temperature, although that assumes that the San Francisco Bay populations are not constrained by being genetically adapted to current conditions here. European green crabs have invaded many places around the world and can tolerate temperatures up to 91° F and reproduce in temperatures up to 79° F (Cohen 2011).

Scenarios 2 and 4 (high sea level rise scenarios): The greatest impact to marsh plain macroinvertebrates will likely be from the conversion of high marsh to low marsh or mudflats due to sea level rise and marsh drowning (if sediment accretion rates do not keep pace with accelerated sea level rise). In most areas around the bay, the area of high marsh and marsh-upland transition zones will decrease as they become compressed between rising sea level and area that lacks upland migration potential, while overall intertidal habitat will likely increase (Strahlberg et al. 2011). Species that depend upon the higher marsh will have less habitat while those that can utilize lower tidal elevations will likely be less affected. Longer inundation periods could be detrimental to some invertebrates but beneficial to others (Robinson et al. 2011), depending on specific life histories. At Toley Creek marsh in San Pablo Bay, tidal inundation did not affect the number of adult insect species or their vertical stratification on marsh vegetation, suggesting that some insects can remain attached to vegetation when inundated (Cameron 1976). Attaching during inundation may be less energetically costly than migrating out of the marsh and back again with the tidal cycle. Many

insects live as adults in the marsh for only a few weeks and so must partition their available energy among development, reproduction, and dispersal in that short time period (Cameron 1976). Reduction in high marsh could reduce host plants available to some insect larva, such as the inchworm moth which lays its eggs on alkali heath (*Frankenia salina*). Loss of their habitat could affect the larval life stage greater than the more mobile adult stage for flying insects. Cameron's study (1976) focused on periodic inundation from tides but could be useful for understanding the possible effects of longer inundation with sea level rise.

Changes in vegetation structure caused either by temperature increases or greater inundation could also affect invertebrate community composition or survival. In a southern California marsh, reduced shade altered the species community and changed the composition of microalgae that provide food to invertebrates, increasing the proportion of insect larvae while decreasing amphipods and oligochaetes (Whitcraft and Levin 2007).

Marsh macroinvertebrates will also need to adapt to changing salinity conditions. Cayan et al. (2008) provide climate projections in which a reduced snow pack and warmer conditions will result in some years with greater runoff (as the result of increased precipitation), and other years with significantly reduced outflow and in particular a shorter runoff season due to reduced late season snow melt. The resulting scenarios indicate increasingly variable salinity in the San Francisco Bay-Delta. Unfortunately, we have little specific information regarding the salinity tolerances of many marsh macroinvertebrates.

Scenario 5 (extreme storm event): Extreme storm events may cause a die-off of vegetation that cannot tolerate increased periods of inundation (Thorne et al. 2013), causing negative impacts on invertebrates living within the vegetation at the time of a storm. This will mostly affect species or life stages that are not very mobile. High storm discharge into tidal marsh creeks could wash away sections of creek banks and displace invertebrates burrowing within them. However, this seems more likely to be a short-term effect on particular marshes.

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## OTHER STRESSORS

The marsh plain may receive inputs of pollutants from adjacent upland areas. Invasive plants could invade the marsh plain, possibly reducing host plants for some insect species, but the impacts would depend on how specialized those insects are on particular plants. Increased frequencies of extreme winter storms could affect everything from salinity to marsh erosion to pollutants washed downstream from uplands.

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## ENTIRE LIFE CYCLE AND INFLUENCES FROM OUTSIDE THE ESTUARY

The effects of climate change may impact different life stages in different ways, with varying degrees of severity. The habitat used by macroinvertebrates, especially insects, often differs among egg, juvenile, and adult stages, as do their tolerance for high temperatures, temperature fluctuations, and other climate-related changes (Kingsolver et al. 2011). However, few data exist on these factors for insects in general and most studies have focused on agricultural pests and pollinators (DeLucia et al. 2012), so it is difficult to extrapolate information for California marshes. Some species may be able to adapt to changes through altered behavior or physiology. In laboratory studies simulating climate change, insects adapted to changed conditions by adjusting their rates of feeding or development (Kingsolver et al. 2009). Invertebrates may also show evolutionary changes to climate change, through selection for particular morphological or physiological traits. Based on global phenology observations of wild plants Wolkovich et al. (2012)

estimates that spring leafing and flowering may advance at the rate of 5-6 days per degree C. These changes in plant phenology may ultimately lead to temporal mismatch with plants and their pollinators (Wolkovich et al. 2012) and insect emergence times with vertebrate consumers (such as juvenile fish, tidal marsh birds, and their young; Dunn et al. 2011).

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## FACTORS THAT MAY AFFECT SPECIES RESILIENCE

Whether marsh macroinvertebrates can adapt to climate change may depend on how the current climate and other abiotic features of the marsh such as inundation rates and salinity compare to each species' tolerance to those stressors. Deutsch et al. (2008) predicted that temperate insect communities would overall see little to some positive effect from increased temperatures, based on the fact that many of these species are already adapted to seasonal fluctuations and not living at the limit of their environmental tolerance. Of course, individual species could still suffer even if overall insect or other invertebrate populations increase.

Invertebrates' survival, growth, and reproduction depend partly on temperature. Species have a range of temperature tolerance and a smaller range of optimal temperature where growth, reproduction, or survival is maximized. Marsh species also experience variation in water and air temperatures that depend on the tidal cycle, freshwater inputs from riverine systems, and by season. A species' resilience to climate change may depend on where current conditions fall within its range of tolerance or optimal conditions. An additional complication is that while a species may inhabit a large geographic range, such as with *Hemigrapsus oregonensis*, local populations may be adapted to local conditions and their resilience to climate change will depend on how well they can tolerate a new range of variation. Finally, the overall effect of climate change on the marsh macroinvertebrate community will also be influenced by interactions among species, such as relative effects on predators versus their prey (Freitas et al. 2007). We lack data on these questions for species in San Francisco Bay marshes so we cannot make specific predictions.

Invertebrates could use behavior to adapt to climate change. Some species might be able to go into diapause to avoid high temperatures. For some species, such as intertidal snails, increased inundation due to sea level rise could reduce stress from desiccation. Others such as aerial insects may need to move higher on vegetation to avoid extended periods underwater. These are educated guesses and also based on the assumption that a variety of micro habitats will remain in sufficient quantities and quality as the sea level rises.

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## LIKELY CLIMATE CHANGE IMPACTS AND RISKS

- Habitat loss due to increased sea levels and inability to migrate marsh landward.
- Possible reduced survival of species and changing abundances and distributions as salinities become more variable under future climate change projections.
- Possible reduced survival of species for which increased temperatures are outside their temperature tolerances but fewer effects and possibly even increased populations of species that are not at their upper limit of temperature tolerance.

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## MANAGEMENT ACTIONS TO BE CONSIDERED

As this group, marsh macroinvertebrates, covers a wide range of organisms, any management action could have positive impacts for some and negative impacts for others. Designing marsh restoration or sea level rise adaptation plans so that future marshes will retain mid- to high marsh elevations and transition zones will help preserve the species and ecosystem functions in the marsh plain. Maintaining or adding connections among marshes, or between marshes and the upland transition zones, would likely benefit many species. See the transition zone chapter for more specific recommendations.

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## UNCERTAINTY AND KNOWLEDGE GAPS

There is much uncertainty about the effects on tidal marsh macroinvertebrates because they have not been well-studied in California. The magnitude of climate change impacts depends in large part on each species' tolerance to changes in temperature or salinity outside the range of current conditions.

Little is known about tidal marsh macroinvertebrates in California and much less is known about the impacts of climate change on these communities. The effects of climate change on macroinvertebrates in general are uncertain, although laboratory studies have provided some examples of what might occur (Deutsch 2008, Kingsolver 2011). Some research has been done in the course of monitoring the progression of marsh restoration projects (see for example Talley and Levin 1999), food webs for fishes and aquatic systems (Howe and Simenstad 2011, Herbold et al. 2014), and changes in food web dynamics due to invasive *Spartina* (Levin et al. 2006, Brusati and Gorsholz 2009). Perhaps the greatest data gap involves impacts to marsh macroinvertebrate distributions, densities, and community structure due to climate change and other environmental stressors and the resulting food web impacts to their vertebrate consumers.

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# Science Foundation Chapter 6

## Carbon Sequestration and Greenhouse Gases in the Baylands

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## INTRODUCTION

Increasingly, but slowly, we as a global society are moving towards linking our land use management, including ecosystem restoration, with activities that reduce our climate change footprint and increase our capacity to respond resiliently at the local level to the challenges of global environmental change. Thoughtful management of San Francisco Estuary's wetlands and Baylands can contribute to local ecosystem resilience as well as play a part in global climate regulation. We also have the opportunity to transfer knowledge gained here, where planning activities have greater support and capacity to be more forward-looking, to other parts of the country and the world.

Incorporating climate change mitigation benefits of wetlands management into planning frameworks is a new focal activity that has developed since the first Goals Report, yet quantification to support these activities in San Francisco Bay has yet to be developed. The management of upland forests for climate change benefits is now well established. Activities on agricultural lands, as found on our Baylands, are being examined to develop best practices to reduce greenhouse gas (GHG) emissions and improve soil carbon sequestration. Over the last 10 years these concepts gradually have moved down slope from drylands to include soil carbon conservation in wetlands, particularly peatlands, and most recently coastal wetlands. The term "blue carbon" has permeated the policy vernacular, reflecting the potential for improved management of coastal wetlands to contribute to climate change mitigation activities, while supporting environmental quality and climate change adaptation.

The management of tidal wetlands is more complicated than forests management, and their areal extent is substantially less; however, the high ecosystem value of coastal wetlands, the greater climate change impact per unit area and the fact that coastal wetlands are being lost a rate greater than any other environment globally and nationally places a particularly high importance on conserving these ecosystems and on finding ways of maintaining them as sea levels rise. Focusing on climate mitigation, activities that reduce or reverse emissions of greenhouse gases, we must factor in impacts of wetlands management on carbon sequestration, methane emissions and to a lesser extent nitrous oxide emissions. In this chapter, we identify key considerations for improving understanding of these processes, considering the extent of historic and current carbon dynamics, as well as possibilities for future processes.

Key questions we address in this chapter are:

- How have carbon and GHG dynamics within the Bay changed over time, and how might they change into the future?
- What role do the Bay's wetlands and Baylands play in greenhouse gas cycling?
- What additional research and data would improve our understanding of carbon and GHG dynamics, and how can this insight improve management of tidal wetlands in the future?
- What are the links between carbon dynamics and other management concerns, e.g., subsidence reversal, sediment management, etc.)?

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## **BACKGROUND**

### **Evolution of a Restoration Policy**

The Bay Area has developed very effective policy and implementation for the conservation and restoration of coastal wetland ecosystems. Early tidal wetland restoration projects were small with isolated impacts, but through progressive experimentation and sharing of experience and knowledge, we have developed projects with rich environmental benefits. Projects have become larger and more complex, incorporated into a coastal mosaic that includes urban areas and agricultural lands. For example, the recent restoration of the Hamilton Wetland on the former Army Airfield beneficially used over six million cubic yards of dredge material from regional maintenance dredging projects and deepening of the Port of Oakland. We also have become one of the leading regions nationally and globally to incorporate sea-level rise projections into coastal wetlands restoration planning. This planning recognizes the need for strategic realignment of coastal infrastructure and sediment management, the importance of restoring early but also the value of projects of different successional ages, and an awareness that some vegetated wetlands will convert to mudflat and open water with sea-level rise, while others in sheltered locations with sufficient sediment inputs will flourish.

As conditions evolve, we are entering a new learning phase, recognizing the potential climate change mitigation (atmospheric GHG reduction) benefits of Baylands and wetlands management by accounting for the changes in carbon storage and fluxes of greenhouse gases. This new phase is at an early stage of development, guided by general principles but limited in detailed quantification. We know that drained wetlands can rapidly release carbon stores that accumulated over hundreds or thousands of years. Drained organic-rich soils may continue to release CO<sub>2</sub> over long periods, and prolonged emissions are evident in drained areas of the Sacramento-San Joaquin Delta, which has been emitting CO<sub>2</sub> continuously for over a century, perhaps as much as 1 billion tons (equivalent to 25% of the carbon in California's standing stock of forests). Conversely, emissions from more mineral-rich soils typically decline or halt over time, and wetland restoration can reinitiate the slow process of carbon sequestration once vegetation is reestablished. In addition to emissions of CO<sub>2</sub>, some wetland soils can release nitrous oxide, N<sub>2</sub>O (a greenhouse gas 310 more potent than CO<sub>2</sub>) and methane, CH<sub>4</sub> (a greenhouse 34 times more potent than CO<sub>2</sub>). Given these substantial greenhouse effects, both N<sub>2</sub>O and CH<sub>4</sub> must be incorporated into any evaluations of overall carbon dynamics and GHG emissions. Nitrous oxide emissions are greatest in wetlands with high nitrate concentrations (e.g., those affected by high fertilizer loads); methane emissions occur in wetlands with

standing water, as well as in drainage ditches and duck ponds, and are more likely to occur at salinities below 18 ppt (~1/2 the salinity of seawater).

A number of actions are ongoing to link overall wetlands management to climate change mitigation responses, both locally and more broadly. California has established a state-level cap and trade system in order to reduce emissions. Early offset projects eligible for credits for reducing GHG emissions within the state have focused on technological solutions. However, in November 2013 the California Air Resources Board (ARB) took the major step of issuing the first eligible carbon credit for compliance offset credit for improved forest management for a coastal region near Willets. Though further behind forestry projects, management of organic soils on drained coastal wetlands and the restoration of these wetlands are being eyed as potential future offset projects. More broadly, carbon market institutions are also exploring the potential to expand their range of activities to recognize wetland management, and in particular coastal wetland management. In 2011 the Verified Carbon Standard (VCS) recognized Wetland Restoration and Conservation activities eligible as potential carbon projects, and recently first global methodology for Greenhouse Gas Accounting Methods for Tidal Wetland and Sea Grass Restoration was submitted for approval (Emmer et al. 2013). In addition, federal agencies have established an interagency team to support blue carbon efforts, and the United States annually reports the official national GHG Inventory, meeting commitments under the UN Framework Convention on Climate Change. Finally, last year the Intergovernmental P on Climate Change (IPCC) provided guidance on incorporating the human impacts to wetlands within accounting for national GHG emissions and reductions (IPCC, 2013).

Whether to support national and state climate change goals, e.g., under a carbon finance framework, or to encourage less formal good practice, there is a need for refined quantification of GHG emissions and removals due to wetlands management at the regional scale. Moreover, wetland climate change mitigation activities should be integrated with regional climate change adaptation strategies to avoid future conflicts in planning outcomes (Crooks et al. in prep).

### Scientific Background

Carbon cycling through plant growth, decomposition, sequestration, and GHG emissions directly affects the sustainability of tidal wetlands. Carbon dynamics have direct links to management concerns for Bay wetlands and are affected by, as well as a feedback to climate change. Tidal wetlands are highly sensitive to sea-level rise but also maintain their balance and remove CO<sub>2</sub> from the atmosphere as they build soil organic matter. In this sense, carbon sequestration within tidal wetlands integrates across both adaptation and mitigation for climate change. Within the Baylands, carbon sequestration is of particular management interest because of the possibility of reversing the loss of elevation due to subsidence, as has been demonstrated on many restoration projects and on a managed subsidence reversal project at Twitchell Island in the Delta, while also providing many other ecosystem services, such as shoreline and flood protection, water quality improvement, and the development of habitat, food web support, and biodiversity.

Organic accumulation within tidal wetlands is strongly correlated with vertical accretion (Turner et al. 2000, Drexler 2011, Callaway et al. 2012, Morris et al. 2012, Schile et al. 2014) and contributes significantly to the capacity of a tidal wetland to build elevation in the face of sea-level rise. Significant stocks of carbon gradually have accumulated within Baylands soils over time, and overall rates of carbon sequestration within tidal wetlands around the world are higher than in many other terrestrial and aquatic ecosystems, particularly when factoring in the continuous burial with sea-level rise (Pendelton et al. 2012). In salt marsh and brackish wetland soils, organic matter commonly contributes up to 20% and 60 %, respectively, by weight in San Estuary (Callaway et al. 2012), with a much greater percentage by volume, given its relatively

low bulk density compared to soil mineral matter (Nyman et al. 1990). As in many tidal wetland systems, the soil carbon content (as a percentage of total soil mass) is generally greatest in low salinity tidal wetlands (freshwater wetlands > brackish wetlands > salt marshes). However, because of the inverse relationship between organic matter content and soil bulk density, the carbon density of wetland soils ( $\text{g C/cm}^3$  of soil) is much less variable, with most wetland soils ranging from 0.02 to 0.04  $\text{g C/cm}^3$  (Gosselink et al. 1984; Chmura, unpublished analysis).

In order for tidal wetlands to remain vegetated, they must accumulate enough mineral and organic sediment to keep pace with on-going rates of sea-level rise. As tidal wetlands accrete sediment, they accumulate carbon within that sediment. Slightly greater rates of sea-level rise will lead to slightly greater rates of carbon sequestration, given adequate suspended sediment concentrations (Morris et al. 2012). However, if wetlands cannot keep pace with sea-level rise and eventually lose elevation, plants become stressed, die, and conversion to unvegetated mudflats occurs, drastically reducing the ability to sequester site-produced carbon (Nyman et al. 1993).

Two critical processes related to carbon sequestration are photosynthesis (conversion of  $\text{CO}_2$  to plant biomass) and decomposition (conversion of soil carbon back to  $\text{CO}_2$ ), with the potential for additional carbon inputs and outputs as either particulate or dissolved organic carbon (POC and DOC). Emission of other GHGs (primarily  $\text{CH}_4$  and  $\text{N}_2\text{O}$ ) also can be important within tidal wetlands. Drainage of coastal wetlands results in the rapid release of  $\text{CO}_2$  to the atmosphere through decomposition of carbon stocks that accumulated over hundreds to thousands of years (Deverel and Leighton, 2010; Crooks et al. 2011, Lovelock et al. 2011, Pendleton et al. 2012), and emissions from drained organic soils may continue for decades until the stock is depleted or management activities change (Drexler et al. 2009, Deverel and Leighton 2010). Reduction of these emissions through management changes and restoration of wetlands is a contributory action to mitigate climate change. There is further potential for reduced vulnerability to climate change impacts, as well-functioning tidal wetlands are effective at reducing storm surge that is likely to increase with on-going climate change.

In addition to  $\text{CO}_2$  that may be sequestered within tidal wetland soils or be emitted from degraded wetland soils, tidal wetlands can emit  $\text{CH}_4$  as a result of sequential oxidation-reduction processes under for anaerobic conditions. As indicated above, methane is an important greenhouse gases, with a much greater greenhouse effect per molecule than  $\text{CO}_2$ . Methane is produced under highly reduced conditions in many wetlands, but due to the high concentration of sulfate ( $\text{SO}_4^{2-}$ ) in waters with a salinity greater than half that of sea water, activity of methanotrophic bacteria is limited (Poffenbarger et al. 2011). Consequently,  $\text{CH}_4$  production is likely to be suppressed in most areas of the Estuary west of Suisun Bay, bar those nedar sources of freshwater or stormwater. Nitrous oxide, by contrast, is primarily produced as a byproduct from human pollution, particularly nitrogen leaching from agricultural fields but also sources in stormwater as well as atmospheric deposition from industry. Estuary and coastal waters are significant sources for nitrous oxide as they recycle nitrogen sources from human activity. Wetlands with permanently wet soils are a minor component of this cycle compared to agricultural soils and open water (EPA, 2010). These emissions are recognized in the development of the US national inventory which applies a default value for  $\text{N}_2\text{O}$  emissions based upon upstream nitrogen application to farm fields (IPCC, 2006)

In the broad geographic scope of the San Francisco Estuary, research on wetland GHG biogeochemistry has been advanced primarily in the Delta, where the majority of former wetland acreage now exists as drained subsided organic soils. Studies have assessed soil carbon stocks (Drexler et al. 2009); rates of  $\text{CO}_2$  emissions with land drainage (Rojstaczer and Deverel 1995); soil carbon accumulation under managed soil building, known as subsidence reversal (Miller et al. 2008); and baseline fluxes of  $\text{CO}_2$ ,  $\text{CH}_4$  and  $\text{N}_2\text{O}$  on drained wetland under a variety of land uses (Teh et al. 2011). However, Delta wetlands are outside of the

scope of the BEHGU project. Within San Francisco Bay, research has focused primarily on wetland response to sea-level rise and restoration (Orr et al. 2003, Stralberg et al. 2011), including some recognition of carbon in wetland building or Holocene response to sea-level rise (Goman 2001), but much less so on CO<sub>2</sub> or other GHG fluxes at annual scales. Data are available for gross soil carbon sequestration rates within natural salt and brackish tidal wetlands (Drexler et al. 2009, Drexler 2011, Callaway et al. 2012), as well as from managed freshwater wetlands (Miller et al. 2008, Miller and Fujii 2010). However, little data are available for sequestration in tidal freshwater wetlands, restored wetlands, or managed wetlands, such as the duck clubs in Suisun Bay. Basic information on soil carbon stocks is limited to analyses of wetland carbon sequestration or to soil survey data collected from drained wetland soils during the 1960s. Beyond these datasets, GHG fluxes in San Francisco Estuary's tidal and drained wetlands are poorly studied. Yet it is this information that is needed to understand the impact of land use, including wetlands restoration, on landscape level GHG fluxes and climate change mitigation. Once basic data on emissions associated with various land uses on the Baylands and for wetland management we will be in a position to more effectively connect climate change mitigation and adaptation for our Baylands system.

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## **APPROACH AND FINDINGS**

Our approach in this section is to draw together available information to improve our understanding of historic, current, and future carbon and GHG dynamics within San Francisco Estuary tidal wetlands (including drained soils in former wetland areas). As with the rest of the BEHGU, a range of future management actions were evaluated, focusing on wetland restoration of Baylands. In our analysis of carbon dynamics and GHG emissions we considered:

- 1) historic emissions of CO<sub>2</sub> by estimating the loss of carbon from historic tidal wetland soils due to the diking and drainage of wetlands;
- 2) on-going carbon dynamics via sequestration and emissions of CO<sub>2</sub> and other greenhouse gases, CH<sub>4</sub> and N<sub>2</sub>O (although specific estimates of current emissions rates were not possible); and
- 3) impacts of restoration on future carbon sequestration and GHG emissions.

Our analyses across all three areas (historic, current, and future carbon dynamics) were restricted by available data, which were quite limited in many cases. No additional field-based data collection was undertaken, and given the shortcoming of available data, we also discuss information gaps as part of our evaluation of management issues. Details on methods are included in Appendix 6.1.

### **Historic Carbon Emissions from Drained Baylands**

Based on the existing soil (Table 6.1 and Figure 6. 1), the overall loss of soil volume due to drainage and diking (see current land cover in Figure 6. 2), and carbon density in these soils (Figure 6. 3), we estimate that approximately 1.2 million metric tons of carbon (4.6 million tons of CO<sub>2</sub>) were released across the region due to tidal wetland drainage over many decades (Table 6.2). It is likely that the majority of these emissions took place in the months and years immediately following drainage because a significant component of the soil carbon that is stored under anaerobic wetland conditions is subject to rapid decompositions when wetland soils are converted to aerobic conditions. However, in areas where soil organic content is very high (e.g., freshwater and low salinity peat soils) emissions may continue for extended periods because of the large pool of organic carbon (Deverel and Leighton, 2010).

We estimated the loss in elevation and soil volume for diked and drained wetlands by assuming historic elevations for tidal wetlands at MHHW (as is the case for most tidal wetlands in the Estuary presently) versus existing elevations in diked areas. The calculated volume losses associated with drainage of wetland area within each segment of the bay, accompanied by estimates of emissions, are provided in Table 6.2.

The South Bay salt pond complex and Central Bay had lower average rates of elevation loss, compared to the North Bay and Suisun Bay; however, because of large areas affected by diking in the South Bay soil loss (13 million m<sup>3</sup>) was comparable to North Bay (20 million m<sup>3</sup>) and Suisun Bay (13 million m<sup>3</sup>; Table 6. 2). Total CO<sub>2</sub> emissions were calculated based on the lost soil volume, along with an average soil carbon density of 0.0233 g C/cm<sup>3</sup> for salt marshes and 0.0334 g C/cm<sup>3</sup> for brackish marshes (based on data from Gosselink et al. 1984, and Callaway et al. 2012), with the emissions patterns across the sub-regions closely mirroring that of overall soil volume loss (Table 7. 2). While these broad estimates of emissions, they give a sense of the scale of total CO<sub>2</sub> flux. By way of comparison, this cumulative emission equates roughly to less than 2 years of ongoing CO<sub>2</sub> flux from soils in the Sacramento San Joaquin Delta (reflecting the high organic content of those soils; Deverel and Leighton 2010).

### Ongoing Soil Carbon Dynamics in the Baylands

Due to the lack of data for the depth of organic soils within tidal Baylands, it is impossible to provide quantitative estimates of carbon stocks in existing tidal wetlands within the Estuary. Data gaps for emissions rates of CO<sub>2</sub> and other GHGs from both natural and diked Baylands, also make quantification of current Bay-wide emissions impossible. Carbon sequestration rates within existing tidal wetlands averaged ~80 g C/m<sup>2</sup>/yr over the last century based on <sup>210</sup>Pb-dated sediment cores from six wetlands across the Estuary including four salt marshes (Whale's Tail in the South Bay; China Camp, Petaluma River Marsh, and Coon Island in the North Bay) and two brackish marshes in Suisun Bay (Rush Ranch and Browns Island; Callaway et al. 2012). There was little variation in sequestration rates within individual wetlands (based on evenly spaced low, mid, and high stations across each of the six wetlands). In addition, variation across sites was also small with only one location (Browns Island) having slightly higher average sequestration rates (107 g C/m<sup>2</sup>/yr). Based on an average value of 80 g C/m<sup>2</sup>/yr and the overall area of tidal wetlands within the Bay (18,200 ha or 45,000 acres), existing wetlands are accumulating an annual total of 14,560 t of carbon across North Bay, South Bay, and Suisun Bay. Although sequestration data are available for mature wetlands within the Estuary, no data exist on rates within recently restored wetlands. Given high rates of sediment accretion in recently restored areas, sequestration rates in these restored wetlands could be higher than natural tidal wetlands over the short term.

In order to provide estimates on existing CO<sub>2</sub> emissions, new soil data are needed, including more comprehensive spatial data on % organic matter, bulk density, soil temperature, depth to water table, and salinity. These data are needed for each soil series, and based on specific land use (i.e., drained vs. natural conditions of the soils). Such datasets would provide the necessary inputs for calculations to estimate rates of subsidence and carbon emissions from soils subsequent to drainage for agriculture, and would form the basis of future simplified, low cost indicators of carbon emissions associated with land use.

In addition to carbon dioxide emissions, methane likely is being released from drainage ditches and areas of standing low salinity and brackish waters on drained Baylands, and nitrous oxide on sites with cattle or subject to nitrogen fertilizer application. Estimates of these emissions could be calculated with some data collection on land use and application of default values provided by the IPCC (2006, 2014). Ideally, the development of regionally specific emissions factors derived from direct measurements would provide more refined analysis of greenhouse gas fluxes within the landscape and climate of the San Francisco Estuary.

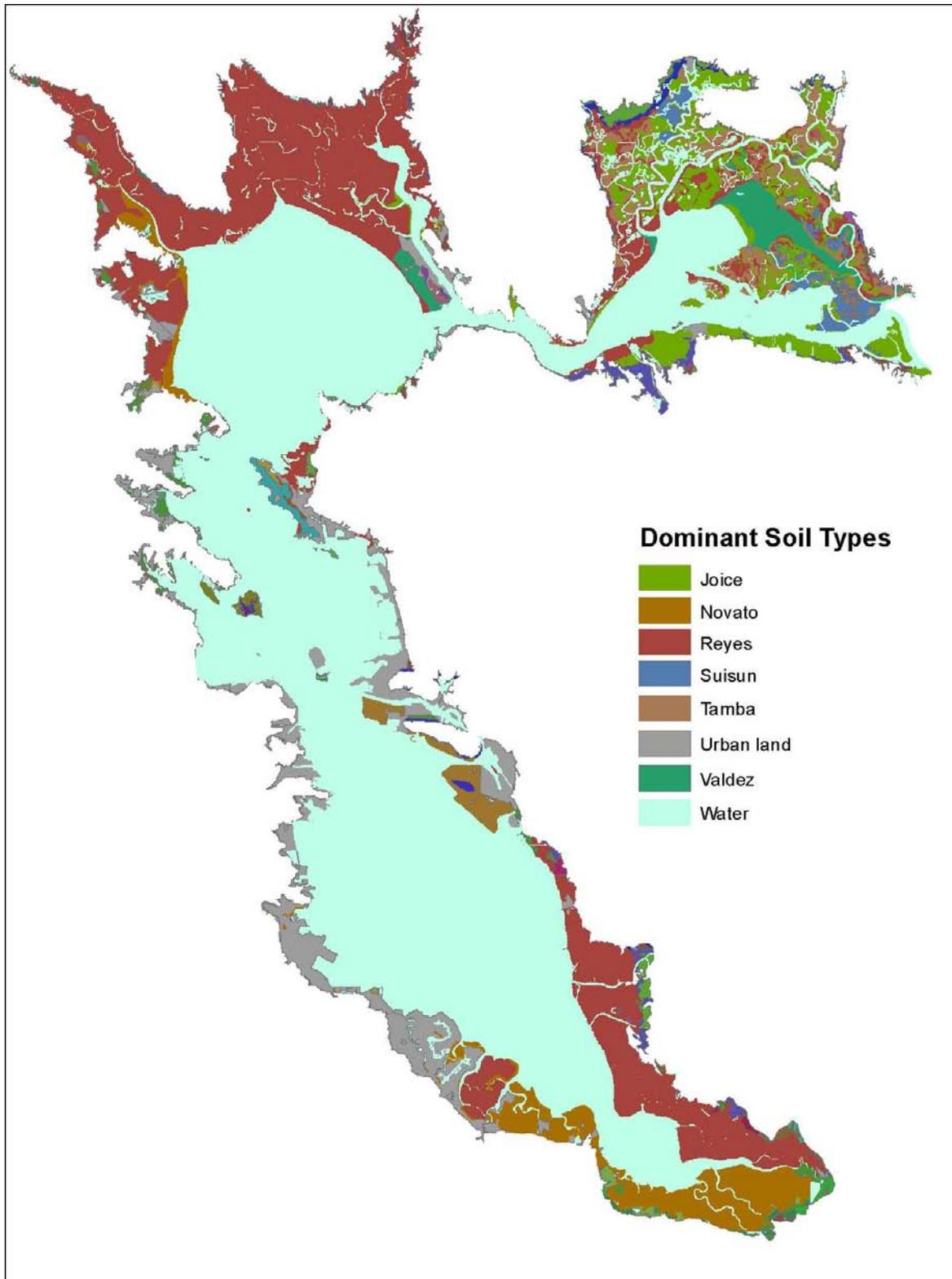
*Baylands Ecosystem Habitat Goals Science Update (2015)*  
*Science Foundation Chapter 6: Carbon Sequestration and Greenhouse Gases in the Baylands*

**Table 1. Acreage of each of the nine dominant soil types classified by current landcover type, region, and segment.**

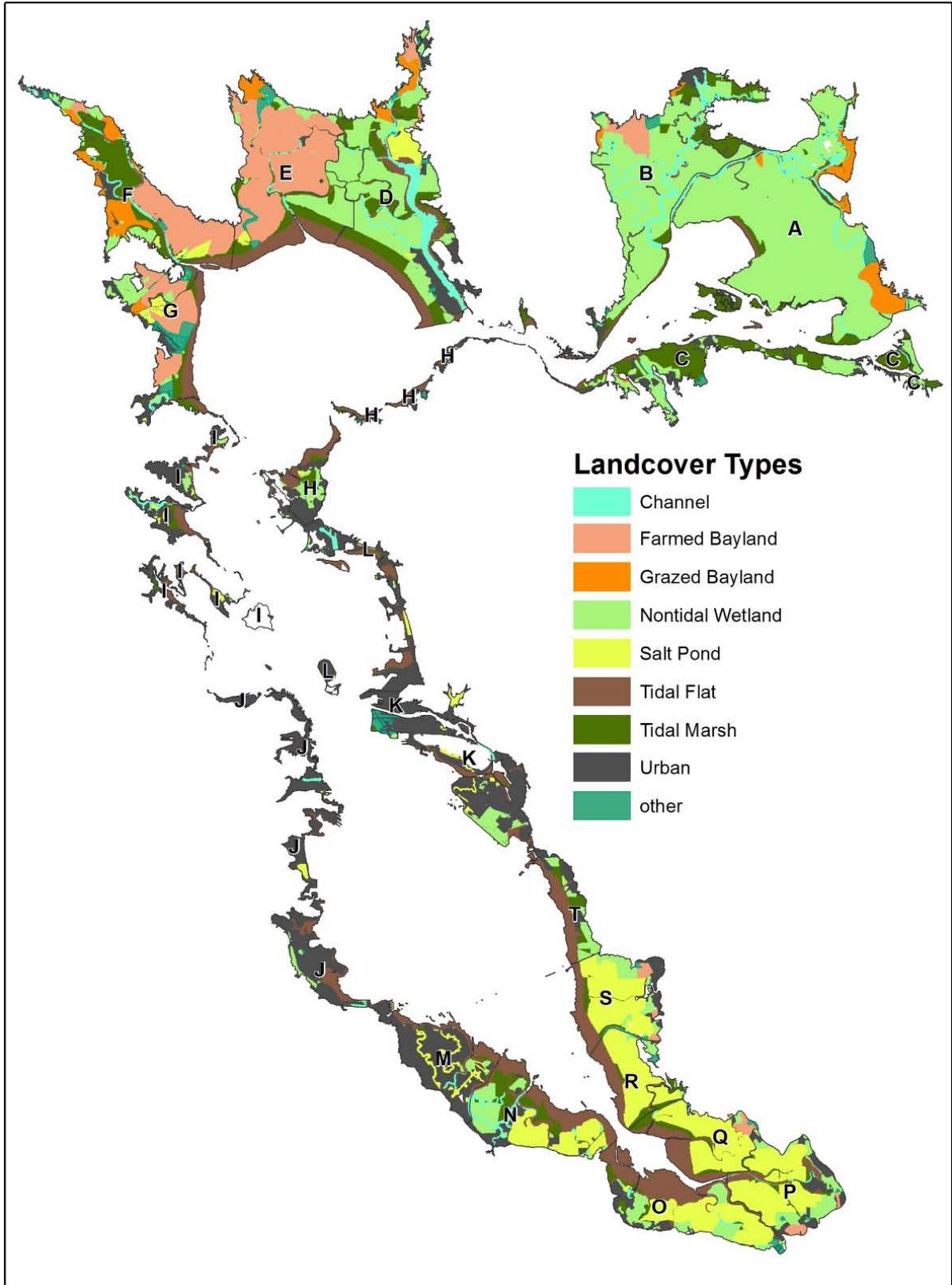
Soil Type	landcover type	Suisun			North Bay					Central Bay				South Bay							
		A	B	C	D	E	F	G	H	I	J	K	L	M	N	O	P	Q	R	S	T
Joice	Channel	46.9	59.9	3.7	12.7																
	Farmed Bayland		18.4																		
	Grazed Bayland	380.2	6.1																		
	Nontidal Wetland	7316.1	4658.3	1224.3	85.8																
	Salt Pond																				
	Tidal Flat	32.0	1.6	4.1	12.9				5.4												
	Tidal Marsh	1502.6	1866.5	3949.9	353.2				17.5												
	Urban	5.0	30.6	273.7	5.5				13.9												
	other	13.5	4.8	13.4	0.4				2.8												
<b>TOTAL</b>	<b>9296.2</b>	<b>6646.2</b>	<b>5469.1</b>	<b>470.5</b>				<b>39.6</b>													
Novato	Channel						12.7	4.6						3.6	3.7	69.0	1.2				
	Farmed Bayland							6.7													
	Grazed Bayland						9.9														
	Nontidal Wetland						387.9	63.0						132.4	100.5	1565.2	631.8				
	Salt Pond							0.7						9.0	2904.1	4002.5	3321.2				
	Tidal Flat						3.1	42.6					11.9	13.1	50.0	280.7	60.4				
	Tidal Marsh						845.8	1149.1					12.0	148.2	1011.8	1215.7	501.7				
	Urban						5.1	45.8					25.4	98.3	272.5	95.0	95.0				
	other						6.2	6.4						0.5	8.2	13.9					
<b>TOTAL</b>						<b>1270.6</b>	<b>1318.9</b>					<b>49.3</b>	<b>405.1</b>	<b>4350.8</b>	<b>7242.0</b>	<b>4611.3</b>					
Omni	Channel			2.0																	
	Farmed Bayland																				
	Grazed Bayland			27.7																	
	Nontidal Wetland			855.5					1.7												
	Salt Pond																				
	Tidal Flat												0.2								
	Tidal Marsh			434.0					1.3												
	Urban			653.4					12.6												
	other			33.4					0.8												
<b>TOTAL</b>			<b>2006.0</b>					<b>16.3</b>													
Reyes	Channel	24.2	46.4	1.1	61.4	39.9	48.6		2.9						15.1		5.5	4.3	1.9		
	Farmed Bayland		334.8		233.6	12456.4	4612.6	3600.4										108.4		112.9	
	Grazed Bayland	506.6	28.7		659.2	616.1	2022.8	139.5													
	Nontidal Wetland	5310.7	4220.9	71.0	9439.7	3133.0	787.6	1590.3	254.0												
	Salt Pond			2.9	1078.6	139.2	347.9	29.5	1.1												
	Tidal Flat	7.0	0.9	23.6	175.1	27.4	20.3	2.7	25.5												
	Tidal Marsh	477.2	599.1	800.4	4953.2	2731.1	3608.2	204.5	499.3												
	Urban	24.3	159.4	98.8	533.2	234.4	494.0	198.8	494.3												
	other	1.3	7.9		2.1	8.0	40.1	9.3	15.5												
<b>TOTAL</b>	<b>6351.3</b>	<b>5394.4</b>	<b>997.9</b>	<b>17136.1</b>	<b>19385.5</b>	<b>11982.1</b>	<b>5774.9</b>	<b>1298.3</b>					<b>42.5</b>	<b>121.7</b>	<b>2816.2</b>	<b>3132.2</b>	<b>6865.5</b>	<b>4270.3</b>	<b>6765.9</b>	<b>2035.5</b>	

Table 1. Acreage of each of the nine dominant soil types classified by current landcover type, region, and segment.

Soil Type	landcover type	Suisun			North Bay					Central Bay				South Bay								
		A	B	C	D	E	F	G	H	I	J	K	L	M	N	O	P	Q	R	S	T	
Suisun	Channel	2.8	11.7																			
	Farmed Bayland																					
	Grazed Bayland	38.1																				
	Nontidal Wetland	3422.4	1148.5																			
	Salt Pond																					
	Tidal Flat		0.1																			
	Tidal Marsh	317.6	86.4																			
	Urban	7.9	12.3																			
	other	5.2	4.4																			
<b>TOTAL</b>	<b>3793.9</b>	<b>1263.4</b>																				
Tamba	Channel	65.1	135.0																			
	Farmed Bayland		236.0																			
	Grazed Bayland	1171.7																				
	Nontidal Wetland	6457.0	4252.3																			
	Salt Pond																					
	Tidal Flat	0.2	11.6																			
	Tidal Marsh	1161.5	769.7																			
	Urban	36.3	122.8																			
	other	4.3																				
<b>TOTAL</b>	<b>8896.1</b>	<b>5527.5</b>																				
Urban	Channel		6.6		6.1		1.1	3.8		24.4	11.3	3.4	4.9	7.5	10.8							
	Farmed Bayland							55.8									7.5					
	Grazed Bayland							80.2	16.4													
	Nontidal Wetland		40.0	105.4	117.8		20.7	449.9	18.8	65.8	208.6	1174.8	36.6	67.1	72.8	2.3	44.8				7.0	
	Salt Pond				5.9		4.6	29.4		38.4	7.6	88.7	64.8	116.5	15.8							
	Tidal Flat		1.2	2.6	5.3		0.2	3.4	6.3	40.8	48.1	40.0	42.6	9.5	1.9	2.3					0.2	
	Tidal Marsh		40.3	47.4	58.7		16.1	31.9	2.2	307.4	47.8	48.9	87.7	82.3	22.5	14.2	1.8				2.8	
	Urban		458.9	472.8	1489.6		287.7	1561.7	621.1	3497.6	8280.4	7989.9	3795.8	6464.8	886.4	59.3	112.7				13.9	210.2
	other		1.3	0.3	17.1			4.3	15.7	123.6	20.2	49.8	74.0	91.6	1.1	17.3						
<b>TOTAL</b>		<b>548.3</b>	<b>628.4</b>	<b>1700.4</b>		<b>410.6</b>	<b>2156.6</b>	<b>664.0</b>	<b>4097.9</b>	<b>8623.9</b>	<b>9395.5</b>	<b>4106.5</b>	<b>6839.2</b>	<b>1011.2</b>	<b>95.5</b>	<b>166.8</b>				<b>13.9</b>	<b>220.3</b>	
Valdez	Channel	6.5																				
	Farmed Bayland																					
	Grazed Bayland	126.2																				
	Nontidal Wetland	6755.6	124.1		906.0																	
	Salt Pond																					
	Tidal Flat																					
	Tidal Marsh	71.9	89.6		389.4																	
	Urban	18.9	3.8		60.1																	
	other	3.0																				
<b>TOTAL</b>	<b>6982.1</b>	<b>217.6</b>		<b>1355.6</b>																		
<b>Grand Total</b>	<b>35319.7</b>	<b>19597.4</b>	<b>9101.4</b>	<b>20662.6</b>	<b>19385.5</b>	<b>13663.4</b>	<b>9250.4</b>	<b>2018.2</b>	<b>4097.9</b>	<b>8673.3</b>	<b>9459.2</b>	<b>4228.2</b>	<b>7244.3</b>	<b>8178.2</b>	<b>7337.5</b>	<b>7910.4</b>	<b>7169.2</b>	<b>4540.8</b>	<b>7272.3</b>	<b>2392.5</b>		



**Figure 6.1.** Distribution of the nine most dominant soil types in the San Francisco Estuary.



**Figure 6.2.** Dominant landcover types surrounding the San Francisco Estuary

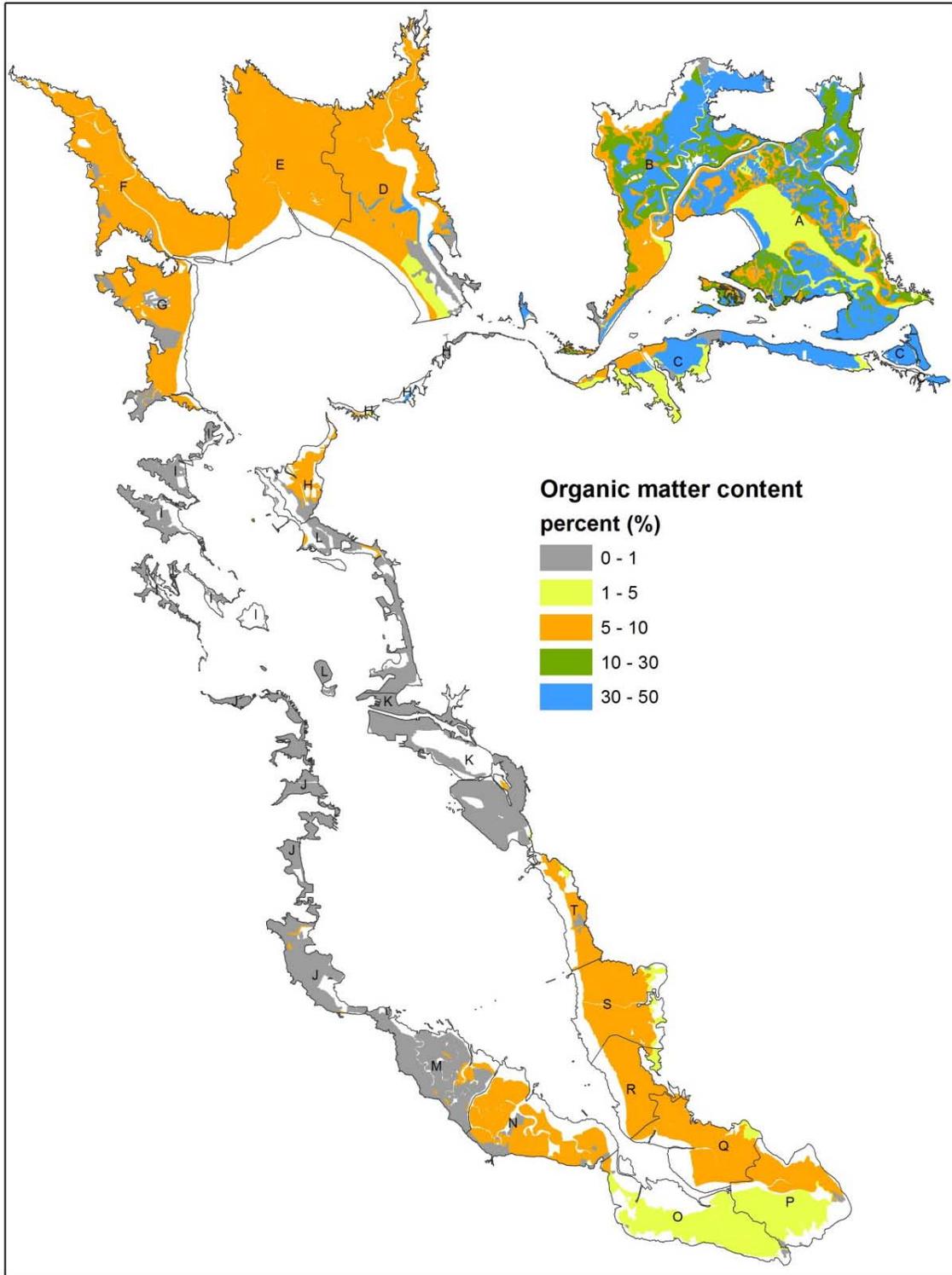


Figure 6.3. Average percent soil organic matter content.

**Table 6.2.** Estimated volume of soil and carbon lost when former tidal wetlands were converted to a nontidal landcover. Non-tidal wetlands are managed or muted wetlands or duck ponds, non-wetlands are areas that have been converted to farming, grazing or urban development, and salt ponds are or previously were non-tidal ponds managed for salt production.

Region	Segment	Volume (m <sup>3</sup> ) Lost by Converted Landcover Type			Total m <sup>3</sup>	Total km <sup>3</sup>	Carbon Lost	
		nontidal wetland	non-wetland	salt pond			metric tons C	CO <sub>2</sub> equivalent
Suisun	A	6,983,130	930,606		7,913,736	0.00791	264,319	968,582
	B	4,522,525	471,990		4,994,515	0.00499	166,817	611,291
	C	250,760	17,475		268,235	0.00027	8,959	32,830
North Bay	D	3,639,040	532,446	211,690	4,383,176	0.00438	102,128	374,243
	E	857,763	6,326,361	12,269	7,196,392	0.0072	167,676	614,439
	F	354,514	5,306,763	230,328	5,891,606	0.00589	137,274	503,035
	G	442,995	2,024,116	186,104	2,653,215	0.00265	61,820	226,536
	H	59,551	8,334		67,884	0.00007	1,582	5,796
Central Bay	I	17,148	118,258	31,323	166,729	0.00017	3,885	14,236
	J	32,960	154,474		187,434	0.00019	4,367	16,003
	K	15,648	504,034	18,721	538,403	0.00054	12,545	45,970
	L	71	7,165		7,236	0.00001	169	618
South Bay	M	117,221	2,053,593	415,976	2,586,791	0.00259	60,272	220,864
	N	814,486	129,703	14,806	958,995	0.00096	22,345	81,881
	O	731,175	333,374	3,096,949	4,161,498	0.00416	96,963	355,315
	P	673,283	1,306,415	1,660,603	3,640,300	0.00364	84,819	310,815
	Q	69,782	340,441	483,584	893,807	0.00089	20,826	76,315
	R	260,924	5,567	8,806	275,297	0.00028	6,414	23,505
	S	302,255	52,720	466,794	821,768	0.00082	19,147	70,164
	T	97,295	15,512		112,807	0.00011	2,628	9,632
<b>Total</b>		<b>20,242,526</b>	<b>20,639,348</b>	<b>6,837,952</b>	<b>47,719,827</b>	<b>0.04772</b>	<b>1,244,954</b>	<b>4,562,069</b>

Despite the lack of data for quantitative estimates, our analysis strongly suggests that, due to the extensive acreage and high organic matter content of its soils, Suisun Marsh is very likely an important source of ongoing CO<sub>2</sub> emissions. Much of San Francisco Bay is likely to be a minor ongoing source as of CO<sub>2</sub> given the low organic content and mineral of the soils, and the substantial length of time since these soils were drained. In addition to being a likely source of CO<sub>2</sub> emissions the diked areas of Suisun Bay are also likely to be a source of CH<sub>4</sub> emissions, emanating from beneath standing water in ditches and duck ponds. In March 2014 an Eddy flux tower was placed on Rush Ranch as a first means to begin to quantify GHG fluxes in that region. (Lisa Windham-Meyers, personal communication).

### Future Soil Carbon Dynamics in the Baylands

Based on projected restoration plans across the bay, a total of 0.28 to 0.30 million metric tons of carbon could be sequestered in restored tidal wetlands across the San Francisco Estuary (Table 6.3). This represents cumulative carbon sequestration across projected restoration projects described in Chapter XX and assumes that carbon would accumulate throughout the rooting depth of tidal wetland plants and across the entire vegetated wetland, with carbon densities reaching values equal to those found in natural tidal wetlands (as above). While this would take multiple decades, it indicates the magnitude of increased carbon sequestration with projected restoration that is possible.

**Table 6.3** Estimated amount of carbon sequestered through current or planned restoration projects using two methods of calculation: GIS differencing and elevation range for vegetation establishment (0.7 to 2.0 m NAVD88).

Segment	Surface Area (m <sup>2</sup> )	former wetland area (m <sup>2</sup> )	% area of segment restored	GIS method		Vegetation range method	
				Volume soil affected (m <sup>3</sup> )	metric tons C	Volume soil affected (m <sup>3</sup> )	metric tons C
A	582739	8251215	0.07	800044	26721	757561	25303
D	1894290	5880508	0.32	1703556	39693	2462577	57378
E	883579	7111593	0.12	807378	18812	1148653	26764
F	563334	4422832	0.13	780685	18190	732334	17063
G	793293	1857552	0.43	1411562	32889	1031281	24029
N	861535	1066977	0.81	979949	22833	1119996	26096
O	1242885	4238253	0.29	2388472	55663	1615750	37656
P	1231506	3358368	0.37	2173095	50633	1600957	37302
S	1442008	2162356	0.67	519928	12114	1874610	43678
T	44137	173472	0.25	27160	633	57378	1337
<b>Grand Total</b>	<b>9539305</b>			<b>11591829</b>	<b>278182</b>	<b>12401097</b>	<b>296606</b>

There are likely to be additional GHG implications of restoring tidal wetlands beyond building carbon stocks that should be recognized. As within the Delta, reducing ongoing emissions may have greater benefits than the actual rebuilding of the carbon stock (Windham-Myers, unpublished data). Reductions in baseline carbon dioxide, methane and nitrous oxide emissions may well occur in Suisun with restoration of wetlands. Potentially there may also be reductions in GHG emissions from drained wetlands west of Carquinez Straits, though as mentioned above the existing soil carbon stock levels in these areas appear depleted.

At the current price of a carbon credit under the California market (approximately \$12 per ton of CO<sub>2</sub><sup>1</sup>) carbon financing would not underwrite the cost of a wetlands restoration project. However, those funds might support existing staff to maintain a science program to provide the monitoring, reporting and verification for crediting. It has yet to be seen what the future price of carbon will be in coming year but given then need for GHG reductions the price is likely to rise.

In order to facilitate restoration and carbon sequestration, a range of novel restoration techniques should be evaluated, including the use of freshwater from waste water treatment plants. Subsidence reversal approaches have been tested in the Delta for over 10 years (Miller et al. 2009, Windham-Myers, unpublished data); however these methods have yet to be applied around the bay. In locations where natural water supplies would be too saline for reed growth freshwater could be derived from redirected waste water outflow. One such opportunity could be the Bel Marin Keys V property in Marin, a highly subsidized potential restoration area immediately adjacent to the Novato Sanitary Districts waste water discharge pipeline. Here expansive areas of low drained Baylands could be converted to managed wetlands, which controlled water elevation management used to rebuild organic soils beneath reed beds. Such approaches could offer a solution to building elevations and filling large accommodation spaces that exist at Bel Marin Keys V and elsewhere. The combined approach of beneficial reuse of waste water, habitat restoration, and rebuilding shoreline elevations would be a useful climate change adaptation strategy in the urbanized setting of San Francisco Bay.

Similarly, restoration of duck pond areas in Suisun would provide multiple benefits, as restoration will likely promote carbon sequestration in recently restored tidal wetlands and simultaneously reduce net GHG emissions. Current management promotes standing water over organic soils, which may reduce current CO<sub>2</sub> emissions and protect soil carbon stocks compared to diked areas; however, this management could increase CH<sub>4</sub> emissions. Restoration of tidal wetlands in this area through dike breaching and subsidence reversal could potentially have a net benefit beyond projected increases in soil carbon sequestration through the reduction of methane emissions; however, further quantification of emission benefits under different land uses is needed.

Other land management options that might be tested include periodic lowering of water levels in the ponds to aerate soils and reduce methane production, as has been explored in rice production (Ma and Lu, 2011). However, halting subsidence through rice production does not address the issues of rising sea level and the combined resulting problems of increases level failure with increased hydrostatic pressure and the reduced restoration opportunity as land surface elevations that progressively fall below that at which vegetation will colonize and tidal wetlands build.

Again, further quantification would aid land use decision making. Subsidence reversal projects, whereby peat accumulation is promoted by impounded surface water, emergent vegetation and substantial accommodation space (e.g., 25 cm depth), may be successful in North Bay locations where climates favor high productivity and low decomposition rates. As per patterns observed in the Delta, CH<sub>4</sub> fluxes may be significant in early stages (Miller and Fujii 2010), and will likely decline with direct or airborne sulfate inputs. Opportunities to apply subsidence reversal organic soils building should be examined for the Baylands. One of our greatest challenges is filling the open volume behind levees created by drainage and subsidence of levees. Subsidence reversal may fill some of that volume.

A further management option might be to manage drained Baylands in such a way as to increase carbon sequestration prior to breaching. Carbon accumulated on the drained Baylands would be stored beneath

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<sup>1</sup> One ton of carbon equates to 3.66 tons of CO<sub>2</sub>

the building wetland. An example of such an activity might include enrichment of soil carbon stocks by conversion to grassland or the application of compost or biochar materials.

An assessment of carbon sequestration on a restoring wetland in the Snohomish Estuary, Puget Sound, documented a breached brackish wetland colonized by soft-stemmed bulrush (*Schoenoplectus tabernaemontani*) to be accumulating carbon at a rate of 352 g C/m<sup>2</sup>/yr (Crooks et al. 2014). Though only a single data point it is comparable with rates of soil carbon accumulation of managed wetlands in the Sacramento San Joaquin Delta (Miller et al. 2009) and a comparable species to that which would colonize restored wetlands in Suisun Bay.

### Data Gaps and Research Needs

As indicated above, major gaps remain in data and understanding of carbon dynamics within Bay wetlands. For example, rates of carbon sequestration in restored wetlands are not quantified, although managed wetlands in the Delta (Miller et al. 2009) and recently restored brackish wetlands in Puget Sound (Crooks et al. 2014) indicate the capacity of restored tidal wetlands to rapidly sequester carbon. We highlight research needs to quantify emissions and sequestration and to improve models of GHG and carbon dynamics:

### Quantification of emissions and sequestration

1. **Measure GHG fluxes (CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O) in wetlands across the salinity gradient and under a range of conditions for nitrogen loading.** We have little data for CH<sub>4</sub> and N<sub>2</sub>O emissions within the Baylands and need to improve estimates of emissions across the salinity gradient. Furthermore, emissions may be affected by nutrient loading, as N<sub>2</sub>O emissions within tidal wetlands are a function of anthropogenic loading.
2. **Measure GHG fluxes and carbon dynamics for disturbed, managed, converted and restored wetlands.** These data would provide fundamental understanding of emissions and removals on drained Baylands or other modified wetlands (i.e., supporting the refinement of GHG emissions for land use activities including modified agricultural practices and wetland restoration). Restored tidal wetlands potentially sequester more carbon than undisturbed natural wetlands in equilibrium with sea-level rise as restored wetlands accrete at rates higher than sea-level rise and rebuild the soil carbon profile.
3. **Improve understanding of the fate of carbon and nitrogen released from eroding tidal wetlands.** If tidal wetlands lose elevation or are eroded, the long-term fate of carbon and nitrogen stored within wetland soils is uncertain. Some portion of the carbon and nitrogen released from the eroding wetlands could return back to the atmosphere, while some could be buried within adjacent subtidal sediments.

### Models, emissions factors and simplifications

We are not yet at the stage of developing process-based or empirical models that enable refined assessment of wetland GHG dynamics or evaluate implications of management activities within the Baylands. Such models would improve GHG management across the landscape and greatly reduce the cost of addressing management questions. Eventually these models could lead to the development of emissions factors, which would greatly simplify emissions and removal predictions. The following modeling advances are needed:

1. **Develop models that describe GHG emissions and reductions with landscape change (e.g., wetland migration, conversion, restoration).** Spatial models exist that describe natural wetland development with sea-level rise across the estuary (Stralberg et al., 2012) and recently were extended to

include dynamic organic matter accumulation rates (carbon sequestration) within natural wetlands (Schile et al., 2014). When data gaps described in the section above have been addressed, additional elements, such as carbon sequestration within restored wetlands, could be added to these landscape models.

2. **Apply process-based models to improve understanding of carbon and nitrogen cycling.** Process-based models (e.g., the DNDC model) require detailed parameterization and have not been applied to Bayland wetlands. Application of these models could improve understanding of GHG emissions and support landscape modeling and management decisions related to Baylands.
3. **Identify simplified monitoring approaches / indicators for a range of emissions-related processes.** Quantification of carbon sequestration by wetlands is an established practice; however quantification across other processes or other types of wetlands is not simple. Based upon empirical quantification and / or model development described above, simplified monitoring approaches could be developed to aid assessment of management activities on drained organic soils or other modified wetlands (e.g., relationships between water table depth, temperature and CO<sub>2</sub> and CH<sub>4</sub> emissions). These simplified approaches would greatly reduce the cost of validation monitoring and model calibration.
4. **Determine default factors of emissions and removals by activity.** Some emissions factors can be developed now based upon existing data (e.g., sequestration of carbon by natural wetlands), some with relatively simple data collection (e.g., carbon sequestration by restoring tidal wetlands) and some will require more detailed quantification of emissions associated with land use (e.g., CH<sub>4</sub> emissions associated with water management on managed wetlands). Once these factors are developed, accounting for GHG emissions and removals across the Baylands will have achieved the same level of understanding that exists for terrestrial lands.

### Recommended Management Actions

In addition to addressing the data gaps above, we suggest that the following management actions be considered:

- 1) Restore wetlands sooner rather than later. This will maximize carbon sequestration by initiating it immediately and increase resilience of restored wetlands to sea-level rise by building wetland elevation.
- 2) Prioritize restoration of wetlands in areas of high sediment availability to promote wetland resilience to sea-level rise and maintain long-term carbon sequestration.
- 3) Explore subsidence reversal and soil building beneath managed wetlands. Suisun Bay is a priority area for these activities, but opportunities exist for test applications on Baylands seaward of the Carquinez Straits. One opportunity is Bel Marin Keys V, as discussed above. Such activities offer the opportunity to increase subsided elevations prior to breaching of dikes, as well as increase carbon sequestration.
- 4) Evaluate activities that can be undertaken on Baylands to reduce ongoing GHG emissions and improve carbon sequestration. These could involve: raising water tables to reduce soil carbon loss from dry soils, filling ditches to reduce CH<sub>4</sub> emissions, and reducing use of fertilizer or cattle densities to reduce soil CH<sub>4</sub> and N<sub>2</sub>O emissions. Additional options include the developing approaches to use of compost to soils from recycled food waste. This may be further enhanced with integrated management of waste water disposal (as currently occurs on the St. Vincent Property). Enhancing soil carbon sequestration

on Baylands would have benefits of their own as well as establishing a pool of carbon that would be buried beneath a restoration project.

- 5) Finally, develop a more detailed plan for prioritizing activities to incorporate climate change mitigation into Baylands management. Developing such a plan would require additional effort beyond this early assessment provided within this chapter.

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## **SUMMARY**

- To understand the climate regulation implications of landscape management strategies, we must compare GHG emissions from the present day landscape (known as the baseline) and compare with GHG fluxes due to management activities.
- Substantial organic carbon was lost from estuarine wetlands when they were diked and drained (approximately 1.2 million metric tons of carbon).
- Organic-rich soils in diked brackish marshes across the San Francisco Estuary are likely to continue releasing CO<sub>2</sub>, while more mineral-rich soils in diked salt marshes are probably depleted of oxidizable carbon. Data are not available to describe the emissions of other GHGs across the landscape.
- Tidal wetlands within the Estuary sequester greater carbon per unit area compared to many other ecosystems, with 14,560 metric tons of CO<sub>2</sub> sequestered annually across all tidal wetlands in the estuary.
- Similarly, data are lacking to estimate GHG emissions from tidal wetlands; however sulfate likely limits CH<sub>4</sub> emissions in high salinity wetlands. No data are available to evaluate N<sub>2</sub>O emissions across the Baylands. Potentially, GHG reductions could be attained by reducing the extent of impounded freshwater behind barriers in Baylands, as well as recognizing the benefits of reducing nitrogen loading to coastal waters from industry and agriculture.
- Restored wetlands are net sequesters of carbon and considering the salinity gradient across the estuary are likely net removers of greenhouse gases over all. It would be helpful to inform management decisions if the fluxes in GHGs across the landscape were quantified.

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**Table 7.J 1** Basic characteristics of soil series in the San Francisco Estuary (Soil Survey Staff, 2012).

<b>Soil Series</b>	<b>% Organic Matter, average</b>	<b>Minimum Depth to Water Table in Summer (cm)</b>	<b>Soil Temperature (C)</b>
Joice	45.0	76.0	17.2
Novato	5.5	46.0	14.0
Omni*	2.5	50.0	16.0
Reyes	6.0	91.0	15.5
Suisun	50.0	76.0	17.0
Tamba	22.5	91.4	17.2
Valdez*	1.3	91.4	16.7

\*Minimum depth to water table only available for drained soil.

**Table 7.J 2** The areal extent of particular soil series in agricultural Baylands.

<b>Soil Series</b>	<b>Agriculture Baylands</b>	<b>Area in km<sup>2</sup></b>
Joice	Farmed Bayland	0.08
	Grazed Bayland	1.56
Novato	Farmed Bayland	0.03
	Grazed Bayland	0.04
Reyes	Farmed Bayland	86.84
	Grazed Bayland	
Suisun	Farmed Bayland	0
	Grazed Bayland	0.15
Tamba	Farmed Bayland	0.96
	Grazed Bayland	4.74

Table 7.J 3 Area of Soil Series Coverage per Sub-region of the San Francisco Estuary (km<sup>2</sup>). Soils series are not indicated within the Central Bay as Bayland areas in this sub-region are all classified as “urban”.

<b>Soil Series</b>	<b>Suisun</b>	<b>North Bay</b>	<b>South Bay</b>	<b>Total</b>
Joice	1.64			1.64
Tamba	5.7			5.7
Suisun	0.15			0.15
Novato		0.07		0.07
Reyes	3.52	98.5	0.9	102.92
<b>Total</b>	<b>11.01</b>	<b>98.57</b>	<b>0.9</b>	<b>110.48</b>

# Science Foundation Chapter 6

## Appendix 6.1: Evaluation of Carbon Dynamics in the Baylands

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In order to evaluate carbon dynamics within historic wetlands within the San Francisco Baylands, it is necessary to evaluate pool and fluxes across the range of different wetland habitats. The historic landscape of the San Francisco Baylands include a rich mosaic of habitat types, with carbon stored primarily in wetland soils that are rich in organic matter. The overall pool of soil carbon within these systems (and current tidal wetlands) is a function of 1) the extent of area covered by tidal wetland habitats, 2) the mass of carbon per unit volume in wetland soils (known as carbon density and determined by soil carbon content and soil density), and 3) the depth of accumulated wetland organic soils. Plant tissues, especially recalcitrant components of belowground biomass, are the dominant carbon source to soil C pools (e.g.,  $\delta^{13}\text{C}$ ) (Malamud-Roam and Ingram 2001). Prior to soil incorporation, above and belowground components of vegetation represent significant albeit short-term standing stocks of carbon.

In estimating pools of soil carbon, we have data on:

- Historic wetland distributions (FROM SFEI/ previous Habitat Goals Project)
- Current soil carbon content and soil bulk density (from soil surveys and recent soil cores); we assume that values from relatively undisturbed sites reflect historic carbon density

However, we do not have any comprehensive and reliable data on the historic depths of wetland soils (or even much data on current depths of wetland soils). Without data on soil depth, it is not possible to estimate the magnitude of carbon pools vulnerable to future emissions; however, historic emissions resulting from changes in pools can be estimated as outlined below.

In estimating emissions of  $\text{CO}_2$  emissions from former tidal wetlands, we have data for:

- Acreage of converted wetlands (including current land use as defined by EcoAtlas)
- Loss in elevation of soil of converted wetlands (assuming that historic marsh plain elevations were close to MHHW and using current elevations of converted wetland elevations from recent LiDAR surveys)
- Current wetland soil characteristics, which can be used as a proxy for historic soil characteristics

To calculate the mass of  $\text{CO}_2$  released with soil drainage, the simplifying assumption is made that the amount of carbon emitted from impacted wetlands soils (kg) is equal to amount of historic carbon throughout that section of the surface soil profile that has been lost = loss in elevation (m)\*area affected by the loss ( $\text{m}^2$ )\*carbon content (as %)\*soil density ( $\text{kg}/\text{m}^3$ ). This first order approximation could underestimate carbon loss by not accounting for additional loss from below the surface of remaining

drained soils, and could overestimate if organic carbon is eroded away with soil and remains deposited in other locations. In estimating historic carbon density and other characteristics of wetland soils, we relied on information from the USDA National Resources Conservation Service SSURGO data (Soil Survey Staff 2013), as well as field data from existing wetland soils. There are seven common soil series found adjacent to the San Francisco Estuary: Joice, Novato, Omni, Reyes, Suisun, Tamba, and Valdez, as well as modified “urban” soils (Figure 1); these eight series cover over 90% of the study area adjacent to the Estuary. We focused our analysis on loss of carbon from wetland soils that have been altered in the past 150+ years by drainage for agricultural activities including row crops, orchards, and grazing, as well as other human impacts, including diking, creation of salt ponds, duck ponds, and urbanization.

Spatially-explicit soil data were downloaded by county and merged for the entire San Francisco Bay area using ArcMap 10.1 (ESRI 2013). The total area of each the eight soil series above was calculated within nine landcover types for each study area segment (Table 1; Figure 2). This area was combined with the change in land surface elevation to calculate the volume of soil lost over the last 150 years in areas that were formerly tidal wetlands but are now diked, muted, or drained. To estimate the loss in land surface elevation, we subtracted current elevations from historic wetland elevations. We assumed that historic wetlands were found at MHHW, and current elevations came from the digital elevation model created by Stralberg et al. (2011). We did not adjust for potential increases in marsh elevations due to increasing sea level over the last 100 years (approximately 20-30 cm), and as a result we may overestimate carbon loss by 10 – 25%. We then calculated the potential loss of carbon through wetland conversion by multiplying the average elevation of each converted wetland by its surface area. Carbon density was estimated from 50-cm sediment cores collected from natural wetlands in North Bay (China Camp, Petaluma River, and Coon Island) and Suisun Bay (Rush Ranch and Browns Island) as part of recent carbon sequestration project (Callaway et al. 2012). Carbon density was calculated based on sediment bulk density ( $\text{g}/\text{cm}^3$ ) and organic carbon content (% carbon). Bulk density was measured directly for 2-cm sections of cores (25 sections/core and approximately 1,000 samples total). Organic carbon content was estimated based on the measured organic matter content (measured by burning subsamples in a muffle furnace [loss-on-ignition method] for the same 1,000 samples) and a correlation between organic matter content and carbon content from a subset of these samples (see Callaway et al. 2012 for details). Carbon density averaged  $23.3 \text{ mg}/\text{cm}^3$  for salt marsh cores and was very consistent across sites. Average carbon density for the brackish sites was  $33.4 \text{ mg}/\text{cm}^3$ , again with little variation between the two sites. An earlier study of Louisiana tidal wetlands found a value of  $26 \text{ mg}/\text{cm}^3$  across a wide range of salt, brackish, and freshwater sites. We used average carbon density values for salt marshes and brackish wetlands,  $23.3$  and  $33.4 \text{ g}/\text{cm}^3$  to estimate carbon loss from historic tidal wetland soils.

To determine the potential carbon storage that could occur with current and planned restoration projects, we utilized a wetland restoration site GIS database compiled by SFEI. We restricted the analysis to restoration sites that are for tidal wetlands only and for those that are in progress or planned. For each restoration site, the volume of soil lost was calculated using the average elevation within the polygon and the surface area of the site. This volume was multiplied by the same carbon densities listed above to calculate the potential carbon storage obtained through restoration.