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Multiple Human and Climate Stressors on California Coastal Marshes and Science-Policy Response

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Multiple Human and Climate Stressors on California Coastal Marshes and Science-Policy Response

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by

Elizabeth Fard

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ABSTRACT OF THE DISSERTATION

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California Coastal Marshes and

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Doctor of Philosophy in Geography

University of California, Los Angeles, 2022

Professor Glen Michael MacDonald, Chair

Coastal wetlands are considered one of the most productive natural ecological infrastructures in the world. Although coastal ecosystems total only 6% of global surface area, they provide an estimated 38% of global ecosystem services. Despite their environmental and societal value, coastlines and coastal habitats are increasingly threatened by human activity. Human threats include proximal disruptions such as wetland removal, changes in sedimentation and chemical pollution. Additionally, climate change, and more specifically sea-level rise (SLR) poses one of the greatest global threats to coastal marshes. Estimates for future SLR rates range anywhere from 0.3 m to over 1.3 m by the end of the 21st century. While historical observations have shown that tidal wetlands can tolerate and dynamically adjust in elevation to some rate of SLR, there are limits. Human population growth, coastal development and the concept of coastal squeeze constrain landward vertical migration of marshes and bring in additional factors that

challenge efforts to understand and manage future salt marsh trajectories. Indeed, humans are integrated into the very fabric of major processes governing wetland stability, which can have major impacts on ecogeomorphic feedback systems and overall marsh resiliency. Thus, local anthropogenic stressors should be coupled with climate change impacts in management and conservation efforts, as they often interact synergistically. However, to do so effectively requires communication and unified actions by stakeholders, managers, and scientists. In the following dissertation, I plan to tie these themes together by researching the multiple human and climatic stressors on California coastal marshes and creating knowledge that can be used in science-policy settings. Furthermore, I use a participant-observer approach to study the communication and planning for mitigating coastal threats in California. First, I obtain high-resolution geochemical data from three coastal marshes in the San Francisco Bay to look at responses to recent anthropogenic changes in sedimentation and pollutant loadings in the context of marsh conditions and histories since the mid-Holocene. Next, I look at attempts to mitigate the impacts of SLR through a large-scale sediment addition project in Seal Beach National Wildlife Refuge, CA. I use the analysis of sediment cores, to understand natural accretion and variability over time and how it compares to the artificial accretion and sedimentation from sediment addition. Lastly, I utilize information from a longer-term participant-observer project updated and augmented with my own participant-observer experience with the SWCASC funded coastal workshops to provide an analysis on knowledge co-production efforts in coastal management settings to understand what makes knowledge relevant in management and policy contexts.

The dissertation of Elizabeth Fard is approved.

Michael E. Shin

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University of California, Los Angeles

2022

For my family – Ali, Manizheh, Anthony, Nima, Houman, and Hourdad

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R. Ambrose, Fard, E., and G.M. MacDonald. (2022). Impacts of climate change on rocky intertidal communities: Research needs. University of California, Los Angeles: Institute of the Environment and Sustainability.

Eppenhimer, D., Fard, E., Kemper, J., Sierks, M., Morrisett, C., Sturtevant, J., Willis, A., Jennings, L. (2021). Climate adaptation planning to support ecosystems and people in the Gila River Watershed, Arizona. University of Arizona, Tucson, AZ, USA: School of Natural Resources and the Environment.

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Paleo to policy: bridging the gap between the paleo-sciences and decision-making, American Geophysical Union (AGU), 2019, San Francisco, CA.

Posters
E. Fard, L. Brown, S. Lydon, G.M. MacDonald. "High-resolution geochemical record of three marshes from the San Francisco bay area". Poster at the 20th International Union for Quaternary Research (INQUA) Congress, 2019, Dublin, Ireland.

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1. Introduction

Coastal wetlands are considered one of the most productive natural ecological infrastructures in the world (Grier et al., 2015; Sun et al., 2017). Thus, they have been extensively researched for decades, covering topics such as sea-level rise reconstruction (Avnaim-Katav et al., 2017), carbon sequestration (DeLaune and White 2012), vegetation mapping (Adam et al., 2010), and modeling (Fagherazzi et al., 2012). Coastal marshes are important habitats for plants and animals that also protect coastal human communities by dissipating wave energy and storm surges (Leonardi et al., 2018). They also absorb and store carbon from the atmosphere (McLeod et al., 2011). Finally, coastal marshes preserve detailed evidence of past climatic, hydrologic, and ecologic conditions in their sediments. These can be valuable archives of past environmental changes and human impacts but can also provide insights into future march responses. Davidson et al (2019) suggest coastal wetlands (only 15% of global natural wetland area) provide \$20.4 trillion per year in ecosystem services. These services impact millions of people, considering that around 38 percent of the population, nearly 2.4 billion people, live within 60 miles of the coast (Neumann et al., 2015).

Coastal wetlands are, however, rapidly disappearing and now represent one of the most heavily used and threatened natural systems in the world (Barbier 2019). This is partly due to the fact that salt marshes have been severely impacted by human activity, and in North America this is particularly associated with European land use over the late 19th and 20th centuries (Silliman et al., 2009; Gedan et al., 2009). In northern California, for example, coastal habitats have been affected by alterations of sedimentation regimes and heavy metal pollution since the gold rush and subsequent industrial boom commencing in the 1850s (Luoma et al., 1998; Ritson et al., 1999). A byproduct of this expansion was an increase of heavy metals and other pollutants produced from mining practices, industrial activity, transportation, municipal waste treatment plants, as well as pesticide and fertilizer use associated with agriculture expansion (Luoma et al., 1998; Monroe and Kelly, 1992). For context, salt marshes are flat and poorly drained areas, subject to flooding, erosion, or alteration for land expansion (Williams et al., 1994). These conditions often lead to coastal zones acting as sinks for heavy metals, which are absorbed and trapped into the sediment as they accumulate (Zhang et al., 2013; Williams et al., 1994). Consequently, following the European period, the increase in urbanization and industrialization led to the buildup of sediments and heavy metal contamination and a decrease in water quality in salt marshes in the region (Hornberger et al., 1999). The changes in sedimentation and the pollutant loading in coastal marshes are important and increasing areas of concern (Li et al., 2022).

While industrial activities have long impacted the performance of these ecosystems, future climate change, and more specifically sea-level rise (SLR), pose a great global challenge to the future of coastal wetlands in the 21st century (IPCC 2022). Historical observations show that tidal wetlands can tolerate relatively high levels of SLR, adjusting via vertical accretion (Kirwan et al., 2016; Holmquist et al., 2021). Over the Holocene, salt marshes have exhibited stabilizing ecogeomorphic feedbacks that allow them to build elevations at similar rates to SLR. Particularly within the temperate zone, ideal conditions such as gradual tidal prism regimes, reliable sediment sources, and accommodation space availability promote salt marsh resiliency. Kirwan et al. (2013) found that SLR can expand tidal networks that allow for delivery of re-suspended sediments to portions of the marsh platform that were previously sediment deficient. Additionally, tectonic activity found along the Pacific can contribute to salt marsh resiliency to SLR by increasing elevation and viability (Patrick and DeLaune 1990), although the presence of tectonic activity may not be enough to offset the impacts of rising sea-levels. These dynamic interactions between hydrology, geomorphology, and ecology allow coastal wetland ecosystems to maintain resilience to climatic changes and disturbances overtime, with relatively consistent recovery rates. The question is – can these processes cope with the rapid SLR anticipated over the 21st century?

Coastal morphology strives to achieve equilibrium as sea-levels rise, which may significantly reshape the coastal landscape. However, recent increases in sea-level rates affect the hydrological regime within the ecosystem which can influence coastal marsh adaptability and resiliency. Estimates for future SLR rates range anywhere from 30 cm up to 1.3 m by the end of the 21st century (Horton et al., 2020), with global scale projections anticipating between 20 and 90 percent of coastal wetland loss (Schuerch et al., 2018). Thorne et al (2018) found that, through the U.S. Pacific region, tidal wetlands are highly vulnerable to end-of-century submergence and loss of overall habitat. Under higher-range SLR scenarios, they found that 83% of current tidal wetlands in the US Pacific would transition to tidal flats by 2110, mostly impacting high and middle marsh habitats. Additionally, low-lying coastal areas can become more vulnerable to rising sea levels through events such as storm surges, tsunamis, and extreme astronomic tides (FitzGerald et al., 2008). Not only can SLR exacerbate these events, but it can also increase the recurrent intervals which may exceed the natural thresholds of the marshes ability to adapt, leading to subsequent drowning (Kirwan et al., 2016; Holmquist et al., 2021). Understanding long-term spatial and temporal changes in SLR impacts is crucial for creating policies and implementing management practices that protect coastal habitats and communities (Wasson et al., 2019).

Growing concentrations of atmospheric CO_2, increased temperatures, melting of glacial ice and thermal warming of the oceans, as well as population growth and the concept of 'coastal squeeze' bring in unprecedented factors that challenge efforts to understand future salt marsh trajectories. As marshes expand landward and reach higher elevations, they begin to border upland forest or grasslands, or in more industrial settings, housing and development (Wiberg et al., 2020). In these upper reaches, sedimentation rates decrease and accumulate mainly though peat accretion (Townend et al., 2011). However, the expansion of coastal cities leaves many temperate coastal marshes bordered by urban development, preventing marsh migration to their adjacent uplands, also known as coastal squeeze (Torio and Chmura, 2013). While there is some level of regional disparity, the increase in urbanization and industrialization in coastal zones causes disruption to the major processes governing wetland stability (Kirwan et al., 2013), which can have major impacts on ecogeomorphic feedback systems and shifting thresholds across the marsh complex. Many of these challenges are exacerbated by climate change impacts, which are projected to occur with the highest 'velocity' in the coastal zone (Loarie et al., 2009). Consequently, the past few decades have seen an increase in coastal wetland research on topics such as coastal marsh formation, resiliency, and conservation.

Despite recent efforts to increase the protection and resilience of coastal marshes, the legacy of centuries of neglect have had long-lasting impacts on these ecosystems. Thus, scientists and managers have started to consider more creative ways of planning and supporting conservation efforts of coastal marshes. One attempt to addressing coastal inundation and erosion, supporting resilient wetlands, has been through the concept of sediment addition. Sediment addition or augmentation projects focus on adding fine sediment dredge materials on top of areas of eroding marsh habitat to increase marsh elevations. This modern collaborative approach has been applied in Essex, UK (Widdows et al., 2006); Venice, Italy (Scarton and Montanari, 2015); Narrow River Estuary, RI (Wigand et al., 2017); along the Mississippi River delta region in southern Louisiana (La Peyre et al., 2009; McCall and Greaves 2022); and at Seal Beach National Wildlife Refuge (NWR) (Thorne et al., 2019). Sediment augmentation efforts are increasingly being utilized as a climate change adaptation strategy to support resilient wetlands. However, due to the mechanical and fiscal constraints of these large-scale collaborative projects, opportunities for implement and execution are few and far between. Marsh characteristics such as geomorphic setting (sediment, topographic), morphology (elevation, vegetation and tidal channel network), inundation frequency (tidal range, SLR and storm activity), and anthropogenic infrastructure influence feedback systems and play a role in landscape reshaping and must be taken into consideration to inform management conservation efforts (Perillo et al., 2019; Kearney and Turner 2016).

While the threats of human land use impacts and climate change, specifically SLR, may often be tractable via scientific studies, the path to incorporating our understanding through science into policy and management settings is not obvious or clear. Research into understanding what makes knowledge relevant in management and policy contexts is not a new field. Mach et al. (2020) argue that scholars and practitioners are eschewing the traditional linear model of knowledge production for a more interactive and engaging model, under the guise of science co-production. Despite the acknowledgements of the benefits of co-creating meaningful knowledge, or actionable science, in decision making contexts, the 'how' of co-production in practice is less clear (Mach et al., 2020; Arnott et al., 2020; Goodrich et al., 2020; Dewulf et al., 2020). There are many reasons for this, and advocates recognize that the large gap in the literature for evaluation of co-production efforts creates an opportunity for contribution to the growing field of actionable science. Still, despite current uncertainties, co-production is advocated as a popular model for creating usable knowledge when compared to the traditional unidirectional flow of information from researchers to policy makers (Mach et al., 2020). Therefore, it is recommended that scientists working at the intersection between research and policy share project successes and barriers to advance thinking on how to create knowledge that is usable and actionable.

This dissertation focuses on two themes: practice and policy. Chapter 2, **High-resolution sedimentological and geochemical records of three marshes in San Francisco, California**, utilizes examination of changing sediment accretion rates, coupled with the longest and highest resolution heavy metal accumulation data from three San Francisco Bay marshes, to examine European impacts in the context of earlier Holocene variability. The results of radiometric dating of sediment cores and geochemical XRF analyses are presented to understand European impacts on the sediments and geochemistry of marshes in the San Francisco Bay over the last ~150-200 years. Chapter 3, **Increasing Salt Marsh Elevation Using Sediment Augmentation: Critical Insights from Surface Sediments and Sediment Cores**, examines the effect of thin-layer sediment application to salt marsh surfaces in Seal Beach National Wildlife Refuge, California, with the goal of mitigating habitat loss caused by accelerated SLR through provision of additional mineral material for elevation gain. I highlight the lessons learned to be able to use this strategy at regular intervals for long term sustainability of Pacific coast marshes in the future. Lastly, Chapter 4, **The politics and economies of coproduction: Assessing the climate science-policy interface in California**, catalogues and assesses efforts to implement science co-production models to manage natural resources in the context of climate change in California, USA, to examine the unintended consequences and politics of the production and circulation of useful science. Additionally, I highlight successes and barriers of science co-production through a principles-focused evaluation framework.

With projected increases in climate change impacts on the horizon, it will be crucial to look at coastal marshes as a whole by incorporating spatial variables and localized biophysical feedback processes in future marsh vulnerability assessments and modeling efforts. Coastal sediments have been widely used in paleoenvironmental research due to their ability to preserve detailed evidence of past climatic, hydrologic, geomorphic, and ecologic conditions (Kennish 2001; Leeper et al., 2017). Understanding trajectories of change in salt marshes over long-term periods provides a more holistic view of when, and more importantly why these ecosystems have changed. Additionally, historical data offer an unparalleled view of natural history that can serve as a powerful tool for understanding future trajectories of change and informing policy (Wasson et al., 2019). Similarly, understanding the different forms and processes for translating science into policy is vital for developing adaptation options and practices that lead to tangible impacts. The data and information in this dissertation will provide an opportunity to look at science co-production through a holistic lens to better understand how to implement and evaluate science co-production in coastal management settings.

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2. High-resolution sedimentological and geochemical records of three marshes in San Francisco Bay, California

2.1 Abstract

The San Francisco Bay has the largest concentration of salt marshes in the state of California. In the last 170 years, the vast majority of the historic tidal wetlands in the Bay have been significantly altered or destroyed due to diking, filling and other processes. Many of the remaining marshes have been impacted by changing sedimentation regimes and related loadings of pollutants such as heavy metals (Sr, Al, Fe, Ti, Cu, Pb, Ni, Zn) over this period, making ecological trajectories and resilience to disturbance uncertain. Here we examine changing sediment accretion rates, coupled with the longest and highest resolution heavy metal accumulation data from three San Francisco Bay marshes, to examine European impacts in the context of earlier Holocene variability. The results are consistent with larger-scale analyses and indicate that European alterations of landscapes and shorelines had geographically variable impacts on marsh sediment accumulation. Despite differential impacts on net sediment accretion, initial European impacts appear to have decreased the proportion of organic material in the marsh sediments, likely due to the delivery of eroded inorganic sediment from landscapes and shorelines. The results confirm significant European impacts on the geochemistry of marshes in the San Francisco Bay over the last ~150-200 years. Post-European arrival levels of Pb are unprecedented throughout the earlier Holocene. However, these values have declined in recent years. Concentrations of Sr, Ti, Cu, Ni, and Zn also increased following European arrival. Our results show that post-European concentrations are not so far removed compared to pre-European maximum concentrations. As sedimentation regimes and emissions of Pb and delivery

of other metals have decreased, and as organic productivity increased on the marshes, environmental trajectories are shifting back towards their immediate pre-European conditions.

2.2 Introduction

Coastal wetlands are one of the most productive and important ecological infrastructures in the world (Sun et al. 2017, Sutton-Grier et al. 2015). They are critical habitats that protect coastal communities, provide refuge to endangered species, absorb carbon from the atmosphere, and preserve detailed evidence of past climatic, hydrologic, and ecologic conditions in their sediments. Davidson et al (2019) suggest coastal wetlands (only 15% of global natural wetland area) provide $20.4 trillion per year in ecosystem services. These services impact millions of people, considering that around 38 percent of the population, nearly 2.4 billion people, live within 60 miles of the coast (Neumann et al. 2015).

Coastal wetlands are, however, rapidly disappearing and now represent one of the most heavily used and threatened natural systems in the world (Barbier 2019). Despite recent efforts to increase the protection and the overall valuation of coastal marshes, the legacy of centuries of neglect have had long-lasting impacts on these ecosystems. In northern California, for example, coastal habitats have been affected by alterations of sedimentation regimes and heavy metal pollution since the gold rush and subsequent industrial boom commencing in the 1850s (Barnard et al. 2013, Gehrke et al. 2011, Ritson et al. 1999, Luoma et al. 1998). Indeed, in San Francisco Bay over the last 170 years, 95% of the historic tidal wetlands in the Bay have been destroyed due to diking and filling, and tidal marsh coverage has reduced to levels less than 4-8% of pre-1850 levels due to European settlement, urbanization of lands, and levee construction on marsh areas (Barnard et al. 2013, Callaway et al. 2011, Ritson et al. 1999, Luoma et al. 1998, Josselyn 1983). Changes in sedimentation, particularly associated with European landcover and shoreline alteration, may produce important changes in marsh topography and extent (Kirwan et al. 2011), and such changing sedimentation and erosional regimes have contributed to geographically variable patterns of marsh erosion or accretion in the Bay (Jaffe and Foxgrover 2006, Foxgrover et al. 2004). It has been estimated that approximately 55% of the sediment load to the Bay in the period from 1849-2011 was due to earlier European alternations of the landscape and that the rates of sediment delivery have declined more recently (Moftakhari et al. 2015).

A byproduct of European alterations and associated sediment loading was an increase in the delivery of inorganic sediments with heavy metals and other pollutants produced from hydraulic mining practices, industrial activities, transportation, municipal waste treatment plants, as well as pesticide and fertilizer use associated with agriculture expansion (Monroe and Kelly 1992). For context, salt marshes are flat and poorly drained areas, subject to flooding, erosion, or alteration for land expansion (Williams et al. 1994). These conditions often lead to coastal zones acting as sinks for heavy metals, which are absorbed and trapped into the sediment as they accumulate (Zhang et al. 2013, Williams et al. 1994). Consequently, following the European period, the increase in urbanization and industrialization led to the buildup of heavy metal contamination and a decrease in water quality in the San Francisco Bay watershed (Hornberger et al. 1999).

Impacts of anthropogenic pollution levels are of major concern to the disruption of biodiversity and ecosystem resilience (Pachauri et al. 2014). Sedimentological records of heavy metal contaminations have been used to provide insights into the chronology of quantifiable levels of anthropogenic pollution (Bhuyan et al. 2017, Zhang et al. 2013, Gehrke et al. 2011, Conaway et al. 2004, Connor and Thomas 2003, Hornberger et al. 1999, Daskalakis and O'connor 1995, Valette-silver 1993). However, the studies of long-term trends of heavy metal pollution in salt marsh sediments are relatively sparse, especially in San Francisco Bay, where many studies focus on the commencement of European settlement and the following century or so (Hornberger et al. 1999, Ritson et al. 1999, Luoma et al. 1998). For example, starting in 1989, the San Francisco Estuary Regional Monitoring Program for Trace Substances (RMP) began recording concentrations of toxic heavy metal contaminants (Ag, As, Cd, Cu, Hg, Ni, Pb, Se, Zn) in the Bay. Results from a 2003 report suggest trace metal concentrations above water quality guidelines, with highest concentrations in the southern region of the Bay (SFEI 2005).

Understanding how European alternations of the Bay watershed and environment impacted sediment properties, accretion and heavy metal concentrations in the San Francisco Bay in comparison to longer-term Holocene conditions provides a more holistic assessment of the true impact of recent human activity relative to natural variability and ecosystem resilience. Although many marshes have been destroyed or highly altered, three marshes at the northern, eastern and southern periphery of the Bay have the capacity to provide long records of sedimentation amenable to such analyses. This study expands upon some earlier work (Callaway et al. 2012) on bay marsh accretion rates by examining long-term, multi-millennial, histories derived from longer cores dated by radiocarbon in addition to ^{210}Pb and ^{137}Cs. We examine long sediment cores from these three marshes to address the questions: 1. How did European land-use and Bay alterations impact sediment characteristics and accretion rates at the marshes; 2. How did European land-use and Bay alterations impact heavy metal concentrations in the marsh sediments; and 3. What are the recent trajectories in marsh conditions relative to their pre-European state. The multi-millennial span of these marsh sediment records allows these questions to be addressed in the context of long-term Holocene states and variability.

2.3 Regional setting

Petaluma Marsh, Browns Island, and Triangle Marsh are part of a small group of San Francisco Bay marshes that preserve continuous sediment records that span from the early Holocene, through initiation of European land use, to the present (Watson and Byrne 2013, Malamud-Roam and Ingram 2004, Watson 2004, Goman and Wells 2000, Ingram et al. 1998). In addition to potentially long sediment records, these marshes were chosen to reflect the major marsh zones of the Bay, specifically the north, eastern and southern edges, respectively (Figure 2.1).

The San Francisco Bay estuary system is the largest estuary in western North America (Luoma et al. 1998). Prior to 10-11 ka, the surficial deposits of the basin were comprised of sands, gravels, and clays deposited by streams. With the close of the Pleistocene, sea level rose approximately 60 m, and filled the lower portion of the basin with soft marine silts, consolidated clay and shale with minor amounts of sand (Josselyn 1983). By 7 to 6 ka, seawater began to flow into the south Bay and the Suisun Basin, bringing with it sediment for the establishment of the late Holocene tidal wetlands and bayward growth of marshes (Josselyn 1983). Following this period of marsh expansion, general trends of freshwater flow, punctuated by periods of high river flows and floods in the estuary, characterize the period between 6 to 4 ka (Goman and Wells 2000). While bay salinity values between 6 to 4 ka were comparable to modern values, estuarine salinity declined between around 4 to 2 ka due to increased precipitation in the Sacramento and San Joaquin watershed (Goman and Wells 2000).

More recently, European settlement impacts have also affected the extent of coastal salt marshes. Specifically in the San Francisco Bay estuary, where historical development and expansion of some marsh habitat have been linked to large amounts of sediment being delivered from land clearing, hydraulic mining debris, and especially agricultural practices such as introduction of livestock and cereal grain harvesting (Watson and Byrne 2013, Hilgartner and Brush 2006, Atwater 1979). This, in some cases, led to increased shallow water areas and marsh expansion. However, the historic extent of San Francisco Bay tidal marshes (220,000 ha) over 150 years ago has been reduced by 90% due to diking and filling (Kirwan et al. 2011, Foxgrover et al. 2004, Williams and Faber 2001, SFEI 1998). As a result, during the last 45 years, extensive efforts to restore tidal wetlands in the San Francisco Bay estuary have taken place (Nagarkar and Raulund-Rasmussen 2017, Williams and Faber 2001).

The study region has a Mediterranean climate, with 94% of annual precipitation occurring from late October to mid-April, and a dry season from May to September with little to no precipitation and cool marine air along the coast and hotter, dry weather inland. Discharge of freshwater into the San Francisco Bay is highest in winter and early spring, owing to direct runoff and snowmelt, influencing the salinity of the Bay. These salinity regimes influence our study sites, characterizing Petaluma Marsh as brackish-polyhaline-hyperhaline, Browns Island Marsh as brackish-oligohaline, and Triangle Marsh as transitional to brackish (Callaway et al. 2012, Shellhammer et al. 2010).

2.3.1 Petaluma Marsh

At the northern portion of the Bay lies the Petaluma River Watershed and Petaluma Marsh, located in southern Sonoma County and northeastern Marin County. It is the largest remaining salt marsh in the San Pablo Bay portion of the San Francisco Bay. The watershed encompasses a 378 km^2 basin and is approximately 30 km long and 19 km wide. The lower 19 km of the Petaluma River flow through the salt marsh. The marsh has an area of ~2023 ha and is surrounded by ~2833 ha of reclaimed wetlands. The two main halophytic plants that dominate the Petaluma salt marsh community are Pacific cordgrass (*Spartina foliosa*) and perennial pickleweed (*Salicornia pacifica*). Precipitation highly influences the flow rates of the Petaluma River, which can experience from 50 to 127 cm of annual rainfall at the highest elevations of the drainage basin (SSCRCD 1999). This can affect the salinity of the marsh and associated soils, which can be very saline during the dry season, and closer to freshwater marsh conditions during the rainy season. Mean tidal range in the marsh is 1.49 m (Callaway et al. 2012). Average annual temperature is 14.3° C, with a range from 7.1° C to 21.4° C (SSCRCD 1999).

2.3.2 Browns Island Marsh

At the eastern edge of the Bay lies Browns Island and its marsh, one of the few remaining tidal-freshwater wetlands in the Sacramento-San Joaquin Delta, with 90% of previous extent being leveed and removed from tidal and floodwater inundation (Reed 2002). The island is a 595 acre natural, tule-dominated, tidal-freshwater wetland, located at the intersection of the Sacramento and San Joaquin rivers. Dominant vegetation at the site are sedges *Schoenoplectus californicus*, *Carex* sp., rushes *Juncus* sp., *Typha* sp., and *Amaranthus* spp. The marsh plain is 1 m above mean low low water (MLLW), experiencing inundation during high spring tides, with a maximum spring tidal range of 1.8 m. Mean tidal range in the marsh is 0.92 m (Callaway et al. 2012). Discharge from the Sacramento River brings higher flows during the winter. Peat formation in the delta region is primarily driven by plant decay associated with freshwater river flows and sediment sources from outside the system (Delusina et al. 2020). Saltwater from the Bay transfers to the delta through the narrow Carquinez Strait, preventing freshwater flows from reaching the Pacific Ocean (Atwater 1979).

2.3.3 Triangle Marsh

At the southern portion of the Bay lies Triangle Marsh, a transitional-brackish marsh, located in Santa Clara County (Shellhammer and Orland 2010). Diking and filing in the southern region reduced tidal marsh area from 225 to the extant 35 km^2 (Watson 2004). The shallow marsh plain, covering 0.35 km^2 and surrounded by salt evaporation ponds, is located at the mouth of Coyote Creek and 4 km from the Guadalupe River. Dominant vegetation at the site are perennial pickleweed (*Salicornia pacifica*) and cordgrass (*Spartina foliosa*). The southern portion of the Bay receives the largest tidal range, with mean tide at 2.1 m and mean spring tide at 2.7 m (Watson 2004). Sediment transport mainly comes from rivers, with highest deposition during winter months associated with peak flows and storm events (Watson 2004, Conomos and Peterson 1977).

2.4 Methods

2.4.1 Field sampling

Sediment cores were collected using a Russian Peat Borer to minimize compression of sediment samples. A 12 meter long sediment core was extracted from Petaluma Marsh (latitude: 38° 10' 1.30" N, longitude: 122° 33' 4.80" W). The first 8 meters of the sediment core were collected in July 2015, and the remaining 4 meters were collected in June 2016. A 10-meter sediment core extracted from Browns Island (latitude: 38° 2' 18.93" N, longitude: 121° 51' 53.26" W) was obtained in 2015. A 6-meter sediment core extracted from Triangle Marsh (latitude: 37° 27' 27.79" N, longitude: 121° 58' 37.20" W) was obtained in 2016. All samples were wrapped in the field, transported back to University of California Los Angeles within a week of collection, and stored in a cold room at 4°C.

2.4.2 Laboratory techniques

2.4.2.1 Chronology

A total of 24 macrofossil samples for radiocarbon dating were taken from varying depths along the cores to develop Holocene chronologies (Table 2.1). A total of 9 dates were obtained from Petaluma marsh, 9 from Browns Island and 6 from Triangle marsh. Roots were not dated. If there are a lack of other datable materials, such as above-ground plant macrofossil material, marine shell samples were dated and corrected for the marine radiocarbon reservoir effect using delta R from Ingram and Souton 1996, for San Pablo Bay (Stuiver et al. 2018, Ramsey 1995). Radiocarbon dating was conducted at the University of California Irvine (UCI) Keck Radiocarbon lab using a 500 kV compact AMS (accelerator mass spectrometer) unit from National Electrostatics Corporation. Plant macrofossil samples and carbonate samples were pretreated following KCCAMS/UCI facilities hydrogen reduction method (Santos et al. 2007). Plant macrofossil organic materials were calibrated using the IntCal20 terrestrial calibration curve (Reimer et al. 2009), and marine shells were calibrated with the Marine20 calibration curve (Heaton et al. 2020).

Cores lengths totaled 1197 cm (12 m) from Petaluma marsh, 1060 cm (10 m) from Browns Island, and 600 cm (6 m) from Triangle marsh. The uncalibrated results from ^{14}C dating of the three cores appears in Table 2.1. For Petaluma marsh, among the nine samples dated, three anomalously young dates (191918, 183280, 191916) were rejected due to extreme stratigraphic reversal. We suspect that the materials dated (both highly preserved sticks) were younger, intrusive organic material that contaminated the sediment during the retrieval process in the field. At Browns Island, nine samples were dated and used. At Triangle marsh, a total of six samples were dated and used. Age-depth models were calculated using the statistical modeling software Bacon 2.2 (Blaauw and Christen 2010).

For chronological control over the past century, radiocesium (^{137}Cs) and radiolead (^{210}Pb) dating methods have been used to improve determination of recent sedimentation rates (Zhang et al. 2013) and thus were utilized in this study. ^{137}Cs and ^{210}Pb chronologies are constructed by gamma spectrometry measurements of radioactivity down a core profile. Generally, profiles are <50 cm in depth, therefore a 2-4 cm^3 sample was extracted approximately every 2-4 cm to a determined depth (~50 cm depth) for each core. This sampling strategy is used to ensure adequate sampling of the prospective ^{137}Cs peak as well as adequate resolution for the ^{210}Pb decay curve. After samples were extracted, they were dehydrated in a drying oven at 110°C for 24 hours and then weighed to calculate bulk density (g cm^{-3}). Samples from Browns Island were ground, sealed in plastic tubes, and sent to the Paleoecological Environmental Assessment and Research Laboratory (PEARL) at Queens University in Kingston, Ontario, Canada, where they were analyzed using a gamma counter for ^{137}Cs and ^{210}Pb activity, using an Ortec high-purity Germanium detector. Samples were prepared following the methods for gamma spectrometry outlined in Schelske et al. (1994). Samples from Petaluma Marsh and Triangle marsh were sealed in plastic tubes and analyzed by gamma counter at the University of Southern California. All samples were allowed three weeks to settle within their sample tubes to obtain a radioactive equilibrium between ^{226}Ra and its decay products which were used to estimate supported ^{210}Pb activity. All raw ^{137}Cs and ^{210}Pb activities can be viewed in the Appendix.

Because we were not able to analyze each interval of the sediment profile for all sites, ^{210}Pb accumulation histories were constructed using the rplum package. This package uses a Bayesian framework to incorporate prior information and more accurately represent uncertainty in the age model than commonly used ^{210}Pb age-depth models, such as the Constant Rate of Supply, and performs well even when there are gaps in the ^{210}Pb profile (Aquino-López et al. 2018). For ^{210}Pb accretion estimates, a Bayesian credible interval is given from the rplum accumulation model along with a mean estimate of accretion for the entire profile. For ^{137}Cs accretion rates, we assume that the year 1963 occurs around the measured peak ^{137}Cs activity and calculate a minimum accretion rate from the ^{137}Cs measurement below the measured peak and a maximum accretion rate from the measurement above the measured peak activity. The mean ^{137}Cs accretion rate is the midpoint of those two possible rates. Age estimates and uncertainties for all ^{210}Pb, ^{137}Cs, and ^{14}C ages were incorporated into a single Bayesian age-depth model using the package rbacon version 2.5.3 with IntCal version 0.1.3 in the R interface (RStudio Team 2020) which is used hereafter in the sediment and geochemical profile analysis.

2.4.2.2 Sedimentological analyses

Sediment cores were sliced into 1 cm intervals. From each slice, a 1 cubic centimeter sample was extracted, dehydrated overnight, burned at 550°C for 4 hours, and at 950°C for 1 hour in order to measure the water content as a percentage of wet weight, bulk density in grams per cubic centimeter, organic content as a percentage of bulk density, and carbonate content as a percentage of bulk density, following standard protocols from Heiri et al. (2001). Remaining material is interpreted to be non-carbonate inorganic sediment component. In this study, loss-on-ignition (LOI) is used to identify bulk density, defined as the mass of organic and mineral components, divided by a wet volume of 1 cubic centimeter (Morris et al. 2016). The sediment material represents locally produced organics and inorganics which are in-washed.

2.4.2.3 Elemental geochemistry analysis

A handheld, battery operated Innov-x Model 2000 XRF Analyzer was used on the entirety of the cores at 2-3cm intervals to detect and quantify elemental concentrations, including Ti, Cr, Mn, Fe, Ni, Cu, Zn, As, Se, Rb, Sr, Zr, Mo, Ag, Cd, Sb, Ba, Hg, Sn, and Pb. The portable X-Ray Fluorescence (XRF) machine was used to obtain elemental concentrations as opposed to wet chemistry elemental analysis techniques, such as acid washes that take several hours and additional resources for preparation. Although normalization calibrations were used to standardize the data set, sample concentrations may appear lower than current concentrations because of the presence of water in the sediments. The concentration of elements measured by the instrument show percent by weight with +/- error estimates. The Compton Normalization method calibration was used to normalize the data. Sample moisture can affect XRF results. However, the Compton Normalization method implemented automatically corrects results for these changes to the sediment matrix. This creates more accurate dry weight results for the samples (Ginau et al. 2019, Olympus 2016).

Of the elements provided in the dataset, Ca, Sr, Al, Fe, Ti, and Si were chosen to provide marine (Ca, Sr) terrigenous and fluvial (Al, Fe, Ti and SI) contextualization (Croudace and Rothwell 2015). The elements Pb, Ni, Zn, and Cu were chosen for characterization of heavy metal pollution based on the findings of Trowbridge et al. (2016) on the concentrations of toxic chemicals in San Francisco Bay sediments recorded from a regional monitoring program in San Francisco. Elemental data are presented as bulk density concentrations rather than fluxes as the latter are sensitive to uncertainties in chronological control, while the former provide informative and robust information on heavy metal loadings.

2.4.3 Numerical analyses

A numerical zonation of the geochemical stratigraphy was developed based on cluster analysis and resulting dendrograms created using the Rioja package software version 0.9-15 (Juggins 2017) in the R Studio interface and Vegan package version 2.4-6 (Oksanen et al. 2018) in R. The Euclidean Dissimilarity Index was the metric distance measure chosen for cluster analysis due to the continuous nature of our dataset. A broken stick model in Vegan was used to determine the optimal number zonation groups within the data. The default agglomeration CONISS method was then utilized for a constrained hierarchical clustering of the distance matrix.

Principal Component Analysis (PCA) is a useful method when exploring multivariate data for reducing the number of dimensions in a dataset by extracting the principal orthogonal components of the data. PCA has successfully been used within geochemical research to extract combinations of variables and explore the relationships within sites (Kirby et al. 2013, Kähkönen et al. 1997, Petterson et al. 1993). PCA was conducted using the Vegan package in R to extract the main components of variability within the geochemical data (i.e., organics, carbonates, XRF) after standardizing and omitting rows with missing values. PCA has successfully been used within geochemical research to extract combinations of variables and explore the relationships within sites (Abu et al. 2020, Kirby et al. 2013, Reid and Spencer 2009). Although using PCA on pXRF data has mainly been used to examine lake sediments, recent research is advancing towards wetland sedimentary analysis (Huang et al. 2016, Mackenzie et al. 2017, Margalef et al. 2013). To further understand the significance of the relationships between the elements and their relationships to organic and carbonate content, a correlation matrix was run on the data. The correlation matrix was computed using the Hmisc and Corrplot packages in R Studio interface.

2.5 Results

2.5.1 Core lengths (cm), chronology and net accretion rates

Cores lengths totaled 1197 cm (12 m) from Petaluma marsh, 1060 cm (10 m) from Browns Island, and 600 cm (6 m) from Triangle marsh. The uncalibrated results from ^{14}C dating of the three cores appears in Table 2.1. For Petaluma marsh, among the nine ^{14}C samples dated, three anomalously young dates (191918, 183280, 191916) were rejected due to extreme stratigraphic reversal. We suspect that the materials dated (both highly preserved sticks) were younger, intrusive organic material that contaminated the sediment during the retrieval process in the field. At Browns Island, nine ^{14}C samples were dated and used. At Triangle marsh, a total of six ^{14}C samples were dated and used. The average modern accretion rates estimated here from the three marsh sites ranged from 2.6–3.8 mm yr^{-1} for ^{137}Cs and from 1.7–2.2 mm yr^{-1} for ^{210}Pb (Table 2.1).^{210}Pb and ^{137}Cs vertical accretion rates determined in this study are similar to vertical accretion estimates from the Callaway et al. (2012) study of six San Francisco Bay marshes that ranged from 2.0–5.0 mm yr^{-1}.

2.5.1.1 Petaluma Marsh

For Petaluma Marsh ^{137}Cs and ^{210}Pb activities were measured on 10 samples. There was a distinct ^{137}Cs peak observed in the ^{137}Cs profile between 16 and 24 cm depth (see Appendix Figure 2.1). While there were particularly low activities of excess ^{210}Pb throughout the core, we were able to successfully construct an accumulation model for the ^{210}Pb using *Plum* for the top 20 cm of the core (see Appendix Figure 2.2a). Petaluma Marsh average ^{137}Cs accretion ranged from 3.1 - 4.6 mm yr^{-1} while ^{210}Pb accretion had a Bayesian credible interval from 1.1 - 3.2 mm yr^{-1}. The mean age estimate given by the ^{137}Cs peak is much younger than that of the age estimates given by the mean ^{210}Pb accumulation model from *Plum*, however, the uncertainty ranges for ^{137}Cs and ^{210}Pb age estimates do overlap. High levels of accumulation can lead to unusually low levels of unsupported ^{210}Pb (Appleby and Oldfield 1978), which may explain the measured ^{210}Pb values seen here. A final *Bacon* accumulation model was constructed using estimated ages and errors from ^{137}Cs, ^{210}Pb, and ^{14}C measurements to resolve some of the disparities between accumulation models and produce a single Bayesian credible interval and mean age estimates for stratigraphic interpretation (see Appendix Figure 2.3a). This final model shows a sharp peak in accretion in the past decades, as might be expected from the low values of excess ^{210}Pb. The estimated age of the Petaluma Marsh core is 6500 cal yr BP.

2.5.1.2 Browns Island Marsh

For Browns Island Marsh 10 samples were analyzed for ^{137}Cs and ^{210}Pb activities. The peak ^{137}Cs activity occurs between 9 and 17.5 cm depth. The *Plum* accumulation model was constructed for the top 12 cm of the core over the past ~70 years. While mean accretion rates are similar for ^{137}Cs and ^{210}Pb, average ^{137}Cs accretion ranged from 1.6 - 3.3 mm yr^{-1} whereas ^{210}Pb accretion had a much wider Bayesian credible interval from 1.0-7.0 mm yr^{-1}. This wide credible interval is due to a high rate of accumulation and wide credible interval prior to a decrease in accumulation over the past decade seen in the *Plum* accumulation model (see Appendix Figure 2.2b). Most probable values of accumulation within the *Plum* model, however, are those which are most similar to the ^{137}Cs accumulation rates. For this reason, the final *Bacon* accumulation model shows a slight slowing of accumulation into the present with a final return to accretion rates similar to those seen over the past few centuries (see Appendix Figure 2.3b). The estimated age of the Browns Island Marsh core is 5650 cal yr BP.

2.5.1.3 Triangle Marsh

At Triangle Marsh, we measured a total of 10 samples for ^{137}Cs and ^{210}Pb activities. The ^{210}Pb profile for this site showed a steady decay in excess ^{210}Pb, but we were not able to detect a peak in ^{137}Cs activity. Difficulty detecting ^{137}Cs in sediments in California is not uncommon, as fallout densities from atomic bomb testing have been shown to be relatively low in California, can migrate throughout the sediment column, and have undergone substantial decay of ^{137}Cs in the time since testing (Drexler et al. 2018). The presence of ^{137}Cs throughout the sediment column at Triangle marsh may be an indication of bioturbation or disturbance at this particular site, but the decrease of total ^{210}Pb activities down the core indicates that bioturbation has not negatively affected the ^{210}Pb profile for this core beyond the capacity for the *Plum* model application (see Appendix Figure 2.2c). From the *Plum* model estimation, ^{210}Pb accretion ranged from 1.2 - 3.7 mm yr^{-1}. This site has a documented history of high accumulation rates due to subsidence (Watson 2004), which is reflected in the high accumulation rate in the ^{210}Pb model over the past 100 years. There is a small plateau in the mid 20th century, followed by accumulation rates increasing once again over the past ~25 years (see Appendix Figure 2.3c). The estimated age of the Triangle Marsh core is 2000 cal yr BP.

2.5.2 Bulk density, organic and carbonate percentage

2.5.2.1 Petaluma Marsh

Bulk density concentrations (Figure 2.2) steadily declined over time since inception of the marsh (approximately 2000 cal yr BP), being at a maximum of 1.12 g cm^{-3} at 6400 cal yr BP. The lowest period of bulk density spanned from 1400-250 cal yr BP, with lowest concentrations of 0.13 g cm^{-3} at 800 cal yr BP. Bulk density during the early post European period rose to 0.68 g cm^{-3}. This is higher than in the immediate pre-European period, but lower than for most of the

Holocene prior to European arrival. In recent decades, bulk density has declined, reaching values of 0.60 g cm^{-3} in the surficial sediments. Organic percent content peaked since the inception of the marsh at 43.6%. Over the Holocene, prior to marsh establishment, organic percent averaged 7.34%. During the early post European period, organic percent declined to 7.25%, lower than in the immediate pre-European period at 13.7%. More recently, organic concentrations increased, being at 12.2% in surficial sediments. Similarly, carbonate percent content peaked since the inception of the marsh at 22.0%. Over the Holocene, prior to marsh establishment, carbonate percent averaged 4.22%. During the early post European period, carbonate percent declined to 3.99%, lower than in the immediate pre-European period at 5.58%. In recent decades, carbonate concentrations increased, being at 4.91% in surficial sediments.

2.5.2.2 Browns Island Marsh

Bulk density concentrations (Figure 2.3) maintained an average of 0.12 g cm^{-3} over the pre-European Holocene, apart from a peak of 0.49 g cm^{-3} at 3750 cal yr BP. Bulk density during the early post European period rose to 0.27 g cm^{-3}. This is higher than in the immediate pre-European period and over the Holocene prior to European arrival. In recent decades, bulk density has again declined, reaching values of 0.16 g cm^{-3} in the surficial sediments. Organic percent content maintained an average of 76.3% over the pre-European Holocene. During the early post European period, organic percent rose to 81.0%. This is higher than in the immediate pre-European period, but lower than for most of the Holocene prior to European arrival. More recently, organic concentrations decreased, being at 57.3% in surficial sediments. Carbonate content is comparatively low at the site, averaging 3.53%, with an unusually high peak at 63.6%, 1150 cal yr BP. The freshwater influence at this site from the Sacramento - San Joaquin Delta may likely contribute to the lower carbonate concentrations.

2.5.2.3 Triangle Marsh

Similarly to Petaluma marsh, bulk density concentrations (Figure 2.4) steadily declined over time, being at a maximum of 1.21 g cm^{-3} at 1800 cal yr BP. The lowest period of bulk density spanned from 460-390 cal yr BP, with lowest concentrations of 0.34 g cm^{-3} at 450 cal yr BP. Bulk density during the early post European period rose to 0.68 g cm^{-3}, which is higher than in the immediate pre-European period, but lower than for most of the Holocene prior to European arrival. In recent decades, bulk density has increased, reaching values of 0.75 g cm^{-3} in the surficial sediments. Conversely, organic percent content has increased over time, being at a maximum of 20.7% at 400 cal yr BP. Over the pre-European Holocene period, organic percent averaged 9.09%. During the early post European period, organic percent declined to 7.56%, lower than in the immediate pre-European period at 9.35%. More recently, organic concentrations increased, being at 10.7% in surficial sediments. Similarly, carbonate percent content has increased over time, being at a maximum of 6.21% at 150 cal yr BP. Over the pre-European Holocene period, carbonate percent averaged 3.43%. During the early post European period, carbonate percent rose to 5.10%, similar to immediate pre-European period concentrations. More recently, carbonate percent decreased, being at 2.68% in surficial sediments.

2.5.3 Dendrogram zones and heavy metal concentrations

Dendrogram, zonation, and cluster analysis (see Figures 2.2, 2.3, 2.4) were used to subdivide the geochemical stratigraphy into discrete zones in order to examine the stratigraphy and relationships between the elements, bulk density, and organic and carbonate content. Vertical distributions of heavy metal elemental concentrations (Ca, Sr, Al, Fe, Ti, Si, Cu, Pb, Ni, Zn) are plotted as absolute concentrations in parts per million (ppm) (Figures 2.2a, 2.3a, 2.4a), as well as ratios of values divided by bulk density (Figures 2.2b, 2.3b, 2.4b) to assess increases in heavy metals independent of increases in bulk materials entering these sites. The concentrations of many of these elements in the deepest parts of the core represent areas without anthropogenic contamination via European industry and land-use practices.

2.5.3.1 Petaluma Marsh

The elemental concentrations vary significantly down core (Figure 2.2a). Importantly, following European expansion, the concentrations of Al, Fe, Si, Cu, Ni, Zn in Zone 1 (240 cal yr BP-present) are not unprecedented within the core, visualized by the overlap of ellipses for Zones 1, and 3-5 (Figure 2.5a). The exception to this is Pb, being at a maximum of 71 ppm at 194 cal yr BP, much higher in the European period (Zone 1) than in other portions of the core. Broadly, Zone 1 shows an increase in all elements following the previous Zone 2. Zone 2 (1300-240 cal yr BP) is characterized by the lowest heavy metal concentrations, relatively. Zones 3 (3150-1300 cal yr BP), 4 (4600-3150 cal yr BP), and 5 (6550-4600 cal yr BP) show natural variations in concentrations over the Holocene, with an overall increase in heavy metals. Zn concentrations peak in Zone 3 at 286 ppm (2500 cal yr BP), while in Zone 5, Ca peaks at 8919 ppm (5750 cal yr BP), Sr peaks at 80 ppm (5950 cal yr BP). Specifically, at 6400 cal yr BP in Zone 5, Al peaks at 45546 ppm, Cu peaks at 93 ppm, and Fe peaks at 92434 ppm.

The elemental ratios share a slightly different story (Figure 2.2b). Zone 1 (300 cal yr BP-present) elements such as Ti, and toxic heavy metals Cu, Pb, and Ni, maintain high concentrations, while Sr, Al, and Si signals are dampened compared to Figure 2.2a. The concentrations of Ca, Sr, Al, Fe, Si, Cu, Ni and Zn in Zone 1, are not unprecedented throughout the core. Importantly, in the marsh establishment Zones 2 (1130-300 cal yr BP) and 3 (1180-1130 cal yr BP), Ca, Sr, Al, Fr, and Si concentrations increase, suggesting changing sediment sources and loads. Zones 4 (1880-1180 cal yr BP), 5 (3150-1880 cal yr BP), and 6 (6550-3150 cal yr BP) show relative consistency in the natural variability of heavy metal concentrations over the Holocene.

2.5.3.2 Browns Island Marsh

In Zone 1 (80 cal yr BP-present) following the European period, elemental concentrations Ti, Si, and the potentially toxic heavy metals Cu, Pb, and Ni are unprecedented throughout the core, visualized by the negative loadings spread of the Zone 1 ellipsis over the PC1 axis (Figures 2.3a, 2.5b). These high concentrations capture terrestrial loadings from the Gold Rush Era (1850s). Zones 2 (1920-80 cal yr BP), 3 (1960-1920 cal yr BP), and 4 (3350-1960 cal yr BP) maintain low heavy metal concentrations over the Holocene. Zone 4 reflects a spike in Fe (16011 ppm), Si (20911 ppm), and Zn (170 ppm) (3350-3300 cal yr BP), following spikes in Ca (17409 ppm) and Sr (162 ppm) in Zone 5 (5630-3350 cal yr BP). Zones 5 and 6 (5650-5630 cal yr BP) reflect natural fluctuations in Al, Fe, Cu, and Zn, possibly capturing changes in terrestrial and fluvial sediment sources.

The elemental ratios suggest that, over the European period, modern increases in bulk density do not bring in higher concentrations of terrestrial heavy metal elements Ti and Ni to the site (Figure 2.3b). Similarly, Fe and Zn signals in Zone 1 (398-0cm) are reduced. Si profile shows dampened modern signals and increased variability throughout the core over the Holocene, possibly capturing changes in detrital input from mechanical weathering of crustal rocks. Still, toxic heavy metals Cu and Pb concentrations are highest in the modern portion of the core following the European period.

2.5.3.3 Triangle Marsh

The elemental concentrations Ca, Sr, Al, Fe, and Si are lowest in Zone 1 (210 cal yr BP - present) (Figure 2.4a). Importantly, following European expansion, toxic heavy metal concentrations of Cu (44 ppm), Pb (72 ppm), Ni (67 ppm), and Zn (112 ppm) are highest in Zones 1 and 2 (343-210 cal yr BP), visualized by the positive loadings and overlap of Zone 1 and ellipses (Figure 2.5c). Elements Sr, Al, Fe, Ti and Si increase in Zone 2, likely reflecting increases in terrigenous input. In Zones 3 (705-343 cal yr BP) and 4 (1927-705 cal yr BP), elemental concentrations of Ca, Sr, Al, Fe, Ti, Si, and Ni show natural variations in concentrations over the Holocene. Zone 4 concentrations of Sr, Al, Fe, Ti, and Si peak during the period prior to European arrival. Comparatively high heavy metal concentrations in Zone 4 are visualized by the higher negative loadings and lack of significant ellipses overlap (Figure 2.5c).

The elemental ratios suggest that toxic heavy metals Cu, Pb, Ni, and Zn are increasing at the site over the European period, especially for Pb, as reflected in Zones 1 (210 cal yr BP - present), 2 (597-210 cal yr BP) (Figure 2.4b). Higher carbonate and Sr concentrations in Zone 1 likely capture marine influence. High concentrations of Al, Ti and Si in Zone 2 and 3 (759-597 cal yr BP) suggest increases in terrigenous input. Zone 4 (778-759 cal yr BP) captures peaks in Si, Al, Fe, Fi, Si, Ni and Zn, potentially reflecting a large terrigenous depositional event. With the exception of Ca and Fe, heavy metal concentration signals are dampened over much of the Holocene, pre-European arrival, in Zone 5 (1927-778 cal yr BP).

2.6 Discussion

2.6.1 How did European land-use and Bay alterations impact sediment characteristics and accretion rates at the marshes relative to longer-term Holocene variability

Comparison of salt marsh trajectories from three sites within the San Francisco Bay shows that salt marsh histories are complex and tied to local hydrologic conditions, with marshes in the delta demonstrating a long, stable record of brackish marsh presence, whereas salt marshes in the north and south bays show much shorter, less-consistent records tied to local hydrology and sediment delivery. The sediments and radiocarbon dates from Petaluma marsh show that the upper salt marsh ecosystem is a geologically recent development. Bayesian-age depth modeling of the stratigraphy shows that while high levels of organics are interspersed throughout the core around 5600 cal yr BP, 4400 cal yr BP, and 3050 cal yr BP, modern marsh establishment occurred around 1900 cal yr BP. Prior to the establishment of the current marsh, the ecosystem most likely resembled a tidal flat with low marsh components or nearby low marsh. At Browns Island, results suggest that the extant marsh exhibited relatively few ecological and geological alterations over the last 6 ka. The ecosystem most likely reflected a consistently organic-rich environment over the Holocene. Results from Triangle marsh show much more variability than Browns Island, and similar variability to Petaluma marsh during the last ~1000 years, specifically, increases in organics and accretion rates increase around 750-500 cal yr BP. At Petaluma marsh and Triangle marsh, sediment conditions were altered in favor of less organic deposits relative to inorganic sediments, likely reflecting increased erosion of land surfaces both proximal and distal to the Bay. Chronological findings are consistent with similar studies tracking marsh expansion in the San Francisco Bay area (Watson and Byrne 2013, Reed 2002, Wells et al. 1997) which concluded that nearby marshes were established between 2000 cal yr BP and 3000 cal yr BP.

Of particular interest are the changes in sediment accretion regimes during European land-use relative to longer-term Holocene variability. Broadly, hydraulic placer mining in the Sierra foothills during the 1850s brought increases in bulk density flows in the form of debris, polluting rivers, creating floods and raising riverbeds (Isenberg 2010). Additionally, anthropogenic impacts from urbanization in the region led to increases in runoff from urban centers and highways, logging, introduction of livestock, cultivation, and municipal plants. These land-use changes have impacts on the delivery rates of sediment to the marsh, which can promote marsh development as well as inhibit marshland expansion. Generally, our results confirm anthropogenic impacts on the San Francisco Bay watershed over the last ~150-200 years following European arrival. However, sediment regimes behave differently at each of the three sites. This finding is inconsistent with a generalized bay-wide paradigm of increases in post-European sediment influx throughout the San Francisco Bay estuary system – the sediment transport, deposition and erosion processes leading to a more complex pattern (Fregoso et al. 2008, Conomos and Petterson 1977). For instance, sediment accretion rates at Petaluma marsh have slightly increased with a large spike at 15 cal yr BP, accretion rates at Browns Island have remained steady with only slight modern increases, while accretion rates have drastically declined and only recently slightly increased over the past half century at Triangle marsh. Generally, there are no long-term increases in accretion over the Holocene at the three sites. However, increases in bulk density are consistent at the three sites. This increase may reflect a switch in sediment source for the marsh, which coincides with a decrease in organic content, resulting from the impacts of European land use. These differences reflect the spatial variability of the sites and how each watershed has responded to anthropogenic influence.

Petaluma marsh and Browns Island, both sites with relatively large watersheds, show increases in bulk density expected from European land-use changes. Petaluma marsh lies at the northern part of the Bay, along the riverbed of the Petaluma River Watershed, near where the river mouth extends into the San Pablo Bay. This location and proximity to the river subjects the marsh to consistent alluvial input, such as river flows and flood events, from the watershed. Browns Island, the eastern-most site of the site, lies at the confluence of the Sacramento and the San Joaquin rivers. The marsh plain is only 1 m above MLLW, so the freshwater flows and river discharge inundate the site annually during high spring tides. Both sites show increases in bulk density and decreases in accretion in the more recent past, likely due to post-European settlement and anthropogenic influences. The increase in bulk density during this period, starting from 300 cal yr BP to modern times at Petaluma marsh and 100 cal yr BP at Browns Island, could be indicative of urbanization and land-use changes leading to increased runoff, soil erosion and increased sedimentation in the watershed post-European settlement.

Additionally, increases in bulk density are coupled with decreases in organic content. At Petaluma marsh, the decrease in organic and carbonate content from 220 cal yr BP is likely a result of changes in sedimentation regimes related to European colonization and subsequent forestry/land-clearance, agriculture, and urbanization. At Browns Island, a site with relatively higher organics and lower mineral sediment, previous studies have concluded that organic accumulation and vegetation changes at the site are tied to minor variations in salinity (Reed 2002). During the early 20th century, periods of drought reduced freshwater input into the rivers and led to the intrusion of brackish water into the marsh region. However, due to the complex agricultural developments along the Sacramento and San Joaquin valleys, efforts such as the Central Valley Project have developed to address water quality issues in the Delta. These alterations may have dampened the increased debris in the form of bulk density, as well as influenced salinity, leading to decreases in organics at the site. However, questions remain regarding whether this decrease in organics is due to increased heavy metal toxicity within the soil composition due to European land use.

A different scenario is observed at Triangle marsh, the southernmost site in the Bay, which shows the largest decrease and subsequent slight increase in sediment accretion in the region over the European period. The marsh plain is shallow and surrounded by salt evaporation ponds. This is partly due to groundwater pumping during the first part of the 20th century in the Santa Clara Valley, which caused subsidence in the southern portion of the San Francisco Bay (Watson 2004). Due to its southern location, the marsh receives sediment mostly through storms and peak flows in the winter months, and high tides in the spring. As the sediments slowly make their way down from rivers, which in some instances can take decades, some sediment may be lost to the ocean. This makes the site susceptible to more localized influences from industrial developments and urbanization. During the 19th century, sediment transport to the south bay was highly influenced by logging in the Santa Cruz Mountains, as well as erosion from logging and grazing from coastal embayment (Watson 2004). Additionally, some of the increases recorded in modern sediment accretion may be due to the increases in organic content at the site. Organic increases may be more locally influenced by fertilization from Santa Clara Valley fields and from inadequate treatment of sewage resulting in added nutrients to the marsh (Grenier and Davis 2010).

2.6.2 How did European land-use and bay alterations impact heavy metal concentrations in the marsh sediments relative to long-term Holocene variability

Similar to sedimentation regimes, our results also confirm anthropogenic impacts on the geochemistry of marshes in the San Francisco Bay over the last ~150-200 years following European arrival. With the arrival of Europeans, erosion in the watershed led to decreased organics as eroded crustal material – including heavy metals – was deposited on the site. Industrial activities, mining operations, and later leaded gasoline centers produced depositions of Pb concentrations higher than any time in the past, when largely only natural processes were at work. From these drivers of patterns, we record post-depositional retention of heavy metals, especially toxic heavy metals, within the salt marsh sediments present at all three sites. Specifically, heavy metal concentrations show an enrichment after European arrival. However, with the exception of Pb, results at Petaluma marsh and Triangle marsh show that modern concentrations of heavy metals are not very different prehistoric conditions, as often suggested by century-scale analyses. While heavy metal concentrations at Browns Island during the European period suggest unprecedented levels, when compared to bulk density inputs, increased levels become dampened for all heavy metals except Pb and Cu. However, with the large decline in the extent of historic wetlands in the Bay following European land use, the remaining salt marsh habitats available for sediment deposition diminished. This change could have resulted in the concentration of suspended sediments and heavy metals in the current marsh network, instead of being a true reflection of an increase in heavy metal pollution (Yellen et al. 2020, Horowitz et al. 2012). But we do record a general decline in heavy metals derived from natural sources in the watershed during the pre-European Holocene.

Interestingly, while heavy metal pollution associated with European arrival is present in the San Francisco Bay, certain heavy metals concentrations are not exceptional when compared to pre-European arrival conditions during the Holocene. Earlier conditions, which reflect natural variability and sediment input from the watershed, suggest that elements such as Ca, Sr, Al, Fe, Ti and Zn are a part of the sedimentological makeup of the soils. For example, terrigenous elements such as Al, Fe, Ti and Si are more abundant in the cores and indicate variations in supply of inorganic sediment to these sites. Additionally, the reducing conditions in estuarine sediments naturally show large quantities of metals, such as iron, which provide the elements necessary for degradation of organic substances in oxic and anoxic conditions. Our results suggest that toxic heavy metals naturally occur in the inorganic sediments within terrestrial environments. Elements such as Cu, Pb, Ni and Zn, however, have low abundance in marine environments. Enrichments of these heavy metals in coastal sediments are generally associated with anthropogenic input. Indeed, results suggest mining operations stemming from the Gold Rush of the 1850s and the subsequent industrial expansion have accelerated sediment yields to rivers in the Bay which have accumulated in the marsh sediments. At Petaluma marsh, higher periods of heavy metal contamination occur pre-marsh establishment when the vegetative platform had not yet developed. At Browns Island, the organic-rich sediments persisted over thousands of years, while heavy metal concentrations were at low or non-existent concentrations prior to European arrival. At Triangle marsh, high organic percent is coupled with lower heavy metal concentrations.

2.6.3 What are the recent trajectories in marsh conditions relative to their pre-European state

Over the last decade or so, most heavy metal concentrations have declined. Indeed, chemical composition of sediments appears to be on trajectories back towards their immediate pre-European conditions. Specifically, heavy metals concentrations of Sr, Fe, Cu, Pb, Ni, and Zn have decreased over the last ~50 years at Petaluma marsh. Similarly, at Triangle marsh, heavy metal concentrations of Sr, Al, Fe, Cu, Pb, Ni, and Zn have declined over the last ~20 years. At Browns Island, heavy metals concentrations Al, Fe, Ti, Pb, and Ni have declined over the last ~20 years. Interestingly, an earlier sediment core study of mercury loadings at Triangle marsh shows a very similar history to what we have revealed for Pb (Conaway et al. 2004). Levels of Hg reached unprecedented concentrations after European arrival but have declined steadily over the latter half of the 20th century. The period of European expansion, and the height of the Gold Rush Era in the 1850s, fostered massive industrial enterprises in the region. The economic demands from the subsequent industrial growth took a toll on the environment, devastating foothills, forests, rivers and estuaries. This exploitation of nature reshaped the environment – one consequence being the heavy metal polluting of river systems and marsh areas in the San Francisco Bay watershed. For example, modern Pb concentrations are seven times greater than Pb concentrations from a century ago (Bruland 1974, Chow et al. 1973). However, at the recognition of the devastation of the environment from these industrial practices, efforts by Californians were made in the late-19th and 20th century to contest and mitigate the destructive effects from these practices (Isenberg 2010). Additionally, due to the decimation of wetlands in the San Francisco Bay during this period, restoration projects have been on the rise in the region (Nagarkar and Raulund-Rasmussen 2016). Thus, environmental improvement can be seen with regards to heavy metal pollution and organic productivity at the three sites. However, the vegetative landscape and soil composition are still different from those prior to European arrival. This suggests that the ecosystem has not fully recovered but may be in a new stage. Indeed, understanding fluctuations in heavy metal concentrations in salt marsh sediments over long-term periods provides a more holistic view of heavy metal variability, the true impact of recent human activity relative to natural variability, and ecosystem resilience to periods of such disturbance.

2.7 Conclusion

The impacts of European alterations of the Bay Area watershed and Bay environments can be tracked in the sediment records from the northern, eastern and southern edges of the Bay at Petaluma marsh, Browns Island and Triangle marsh, respectively. Sediment conditions were altered in favor of less organic deposits relative to inorganic sediments, likely reflecting increased erosion of land surfaces both proximal and distal to the Bay. Net accretion rose slightly at Triangle marsh and at Browns Island, likely due to increasing amounts of sedimentary material, but net accretion fell at Petaluma Marsh, likely due to deceased marsh vegetation productivity. The net accretion rates during the European period were lower than natural variability in accretion during the earlier Holocene at Petaluma marsh and Browns Island, but higher at Triangle marsh. Toxic heavy metals including Cu, Pb, Ni and Zn increased at all marshes, likely due to increased erosion of crustal materials with native metal concentrations and due to emissions of metals from mining, industrial and transportation activities. However, with the exception of Pb, none of the concentrations of these metals in the post-European exceeded maximum concentrations observed in the earlier Holocene during periods of largely inorganic sedimentation at Petaluma marsh and Triangle marsh. At Browns Island, European period concentrations of Cu, Pb, and Ni are unprecedented. Over the past few decades the trajectory of a number of these indicators, including notably Pb, is returning to conditions more typical of the immediate pre-European period. Although these trajectories suggest resilience and return to more natural ecological conditions, how anticipated 21st century sea level rise will impact these marshes remains an important concern (Pachauri et al. 2014, Parker et al. 2011).

2.8 Figures

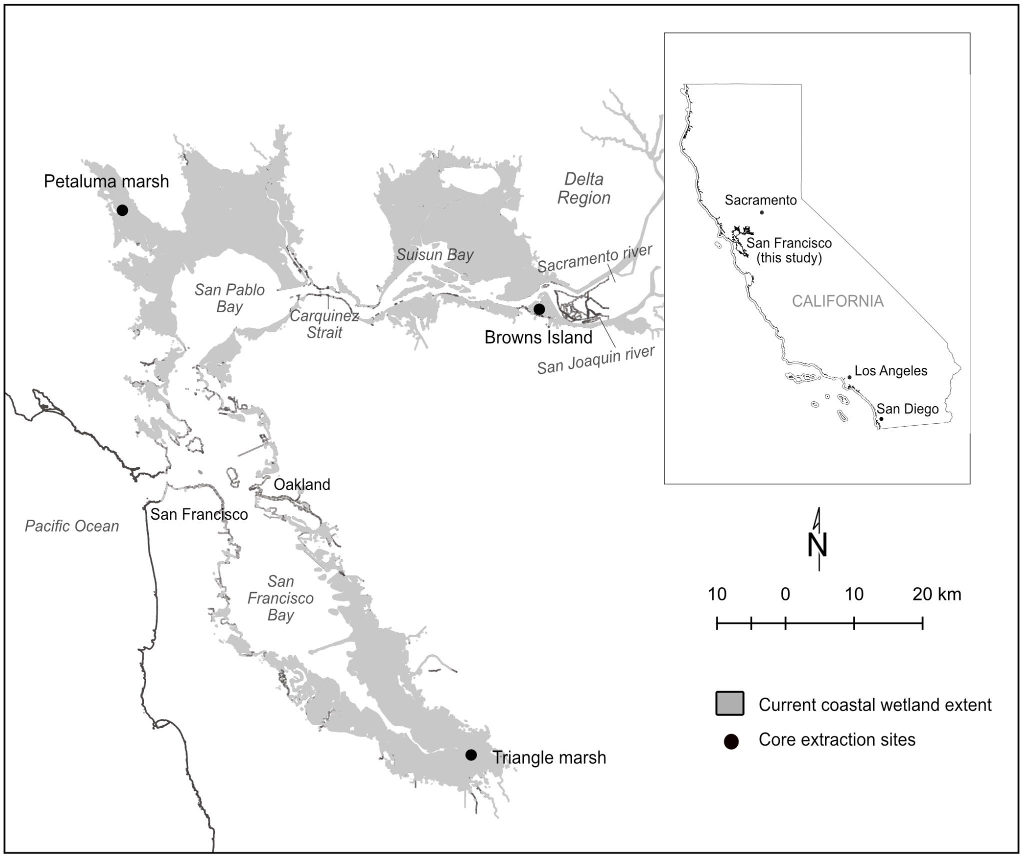


Figure 2.1 Site map of San Francisco Bay marshes

Site map of core extraction locations. Petaluma Marsh (PTL, latitude: 38° 10' 1.30" N, longitude: 122° 33' 4.80" W), Browns Island (BI, latitude: 38° 2' 18.93" N, longitude: 121° 51' 53.26" W), Triangle Marsh (TRM, latitude: 37° 27' 27.79" N, longitude: 121° 58' 37.20" W).

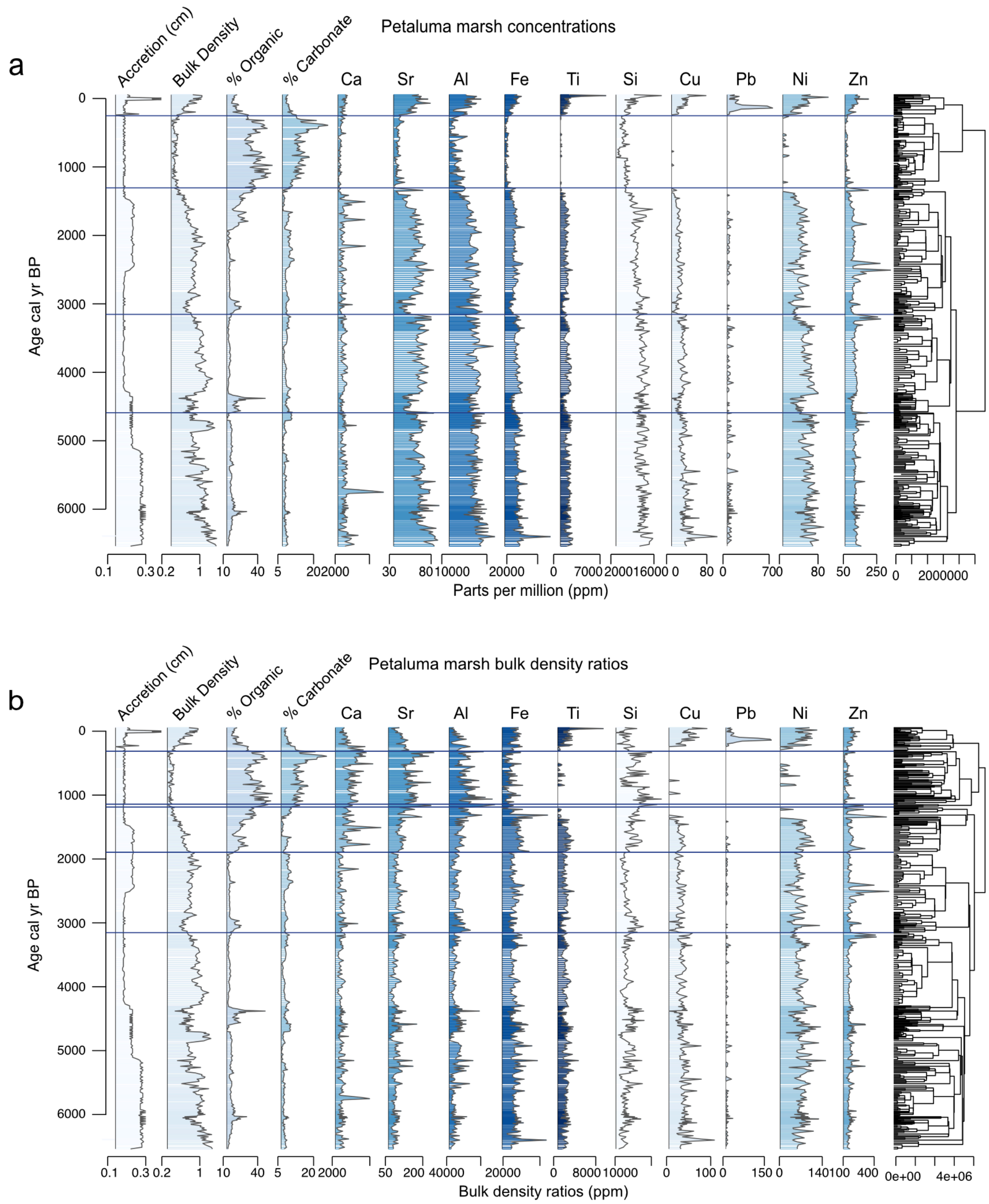


Figure 2.2 Stratigraphic profile for Petaluma Marsh

Petaluma Marsh (PTL). a) Stratigraphic profiles of heavy metal concentrations in parts per million (ppm), bulk density, organic content and carbonate content; b) stratigraphic profiles of heavy metal concentrations divided by bulk density. Rioja package in R was used for zonation, cluster analysis, and dendrogram. Age is given in cal years BP from the BACON model.

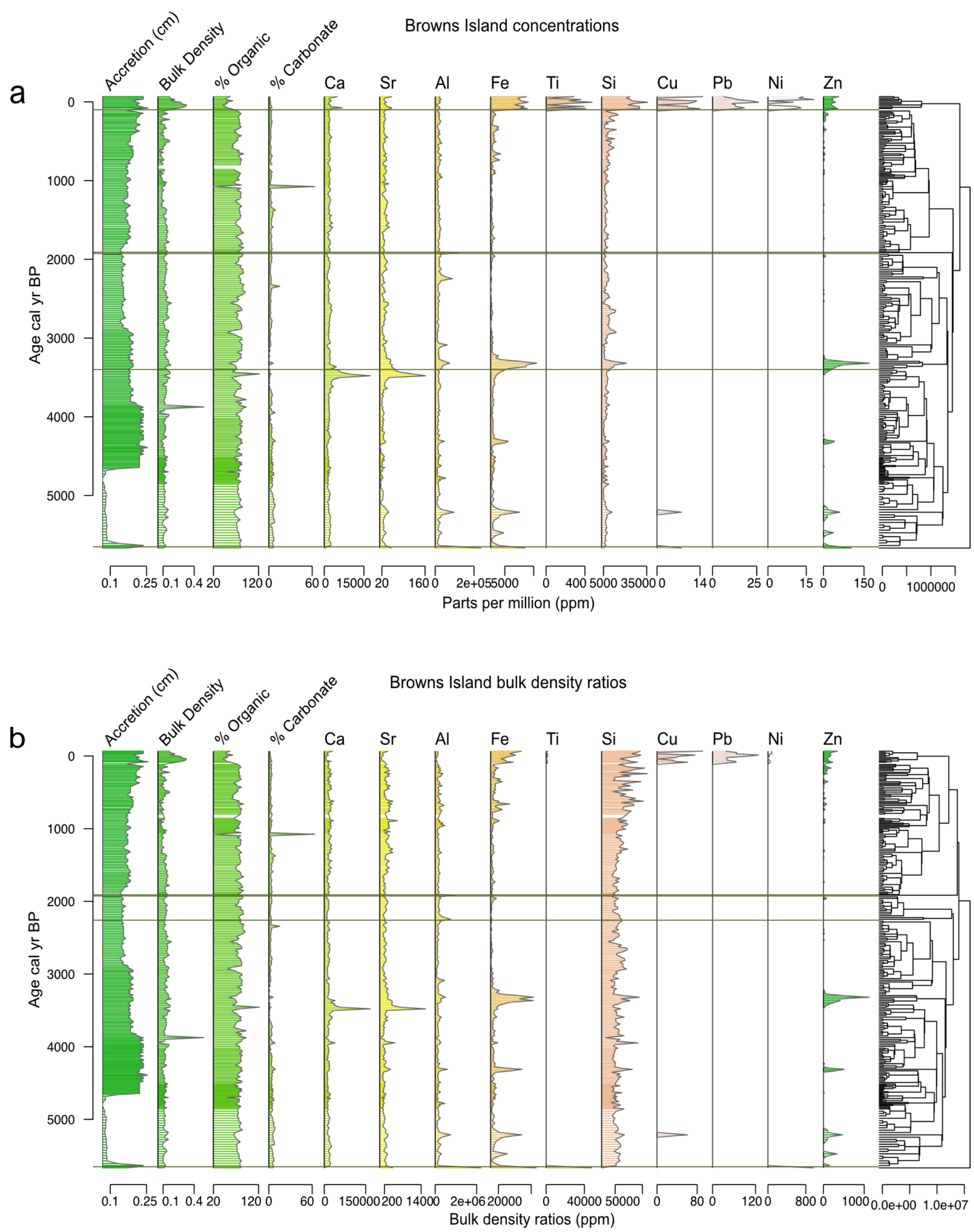


Figure 2.3 Stratigraphic profile for Browns Island Marsh

Browns Island Marsh (BI). a) Stratigraphic profiles of heavy metals concentrations in parts per million (ppm), bulk density, organic content and carbonate content; b) stratigraphic profiles of heavy metal concentrations divided by bulk density. Rioja package in R was used for zonation, cluster analysis, and dendrogram. Age is given in cal years BP from the BACON model.

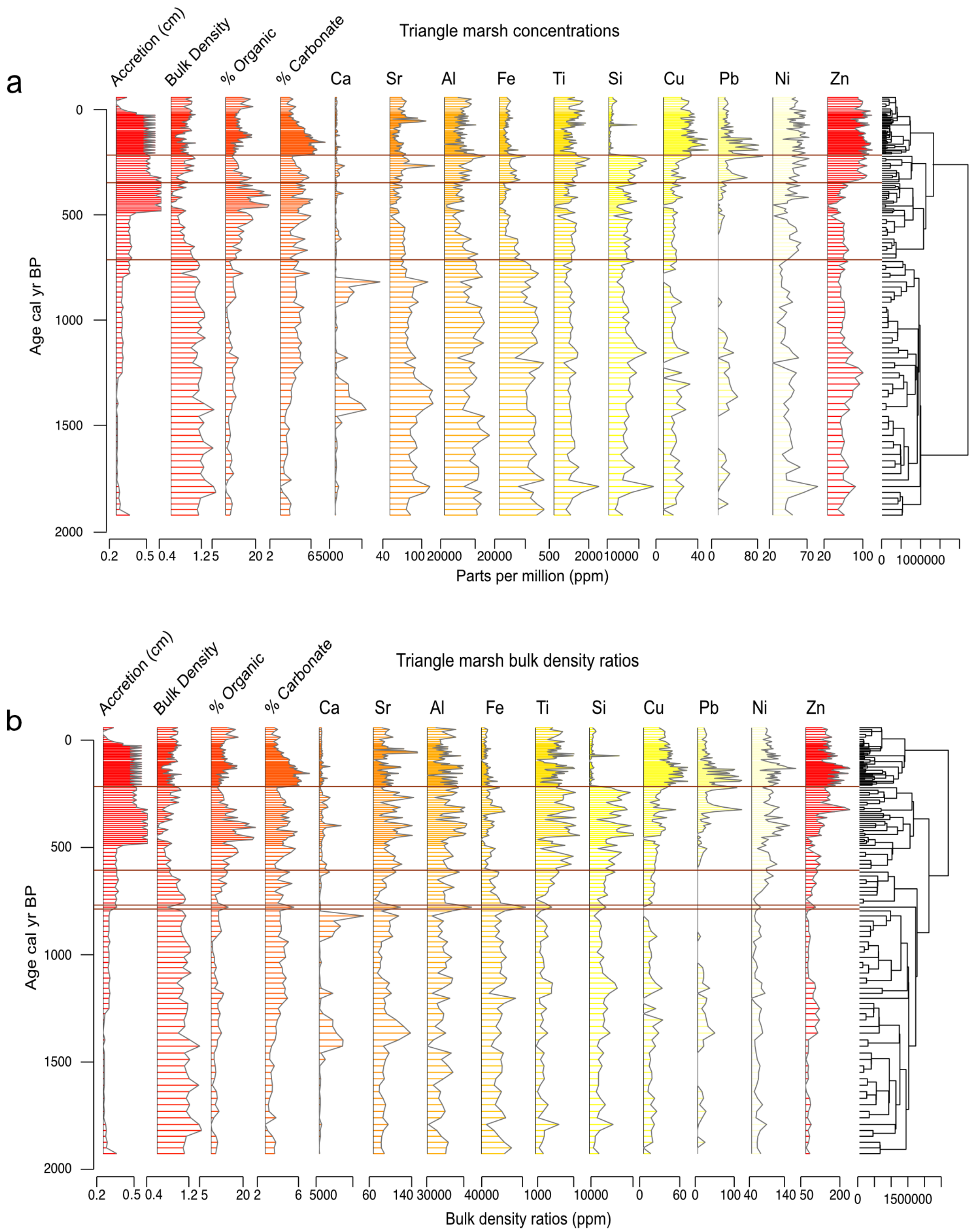


Figure 2.4 Stratigraphic profile for Triangle Marsh

Triangle Marsh (TRM). a) Stratigraphic profiles of heavy metals concentrations in parts per million (ppm), bulk density, organic content and carbonate content; b) Stratigraphic profiles of heavy metal concentrations divided by bulk density. Rioja package in R was used for zonation, cluster analysis, and dendrogram. Age is given in cal years BP from the BACON model.

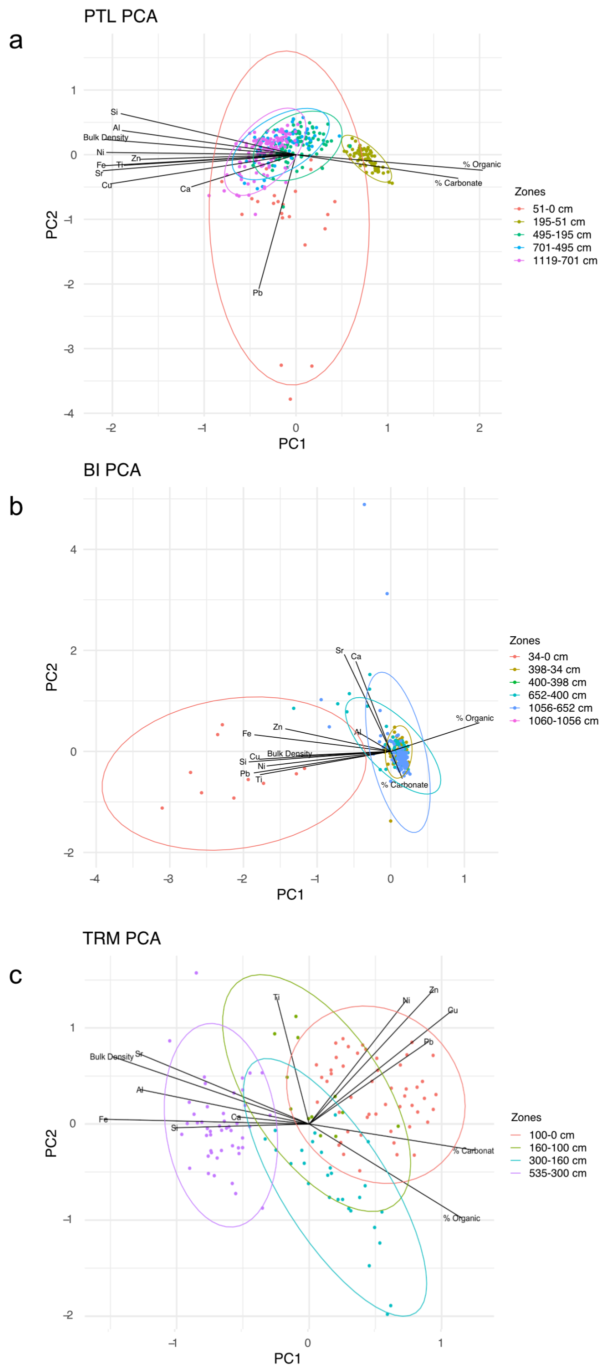


Figure 2.5 PCA analysis of variables

Principle component analysis of XRF heavy metal elements, bulk density, organic content and carbonate content by depth (cm). Data were analyzed and graphed in RStudio. a) The two axes explain 71.5% of the variance at Petaluma Marsh. b) The two axes explain 58.7% of the variance at Browns Island Marsh. c) The two axes explain 60.3% of the variance at Triangle Marsh.

2.9 Tables

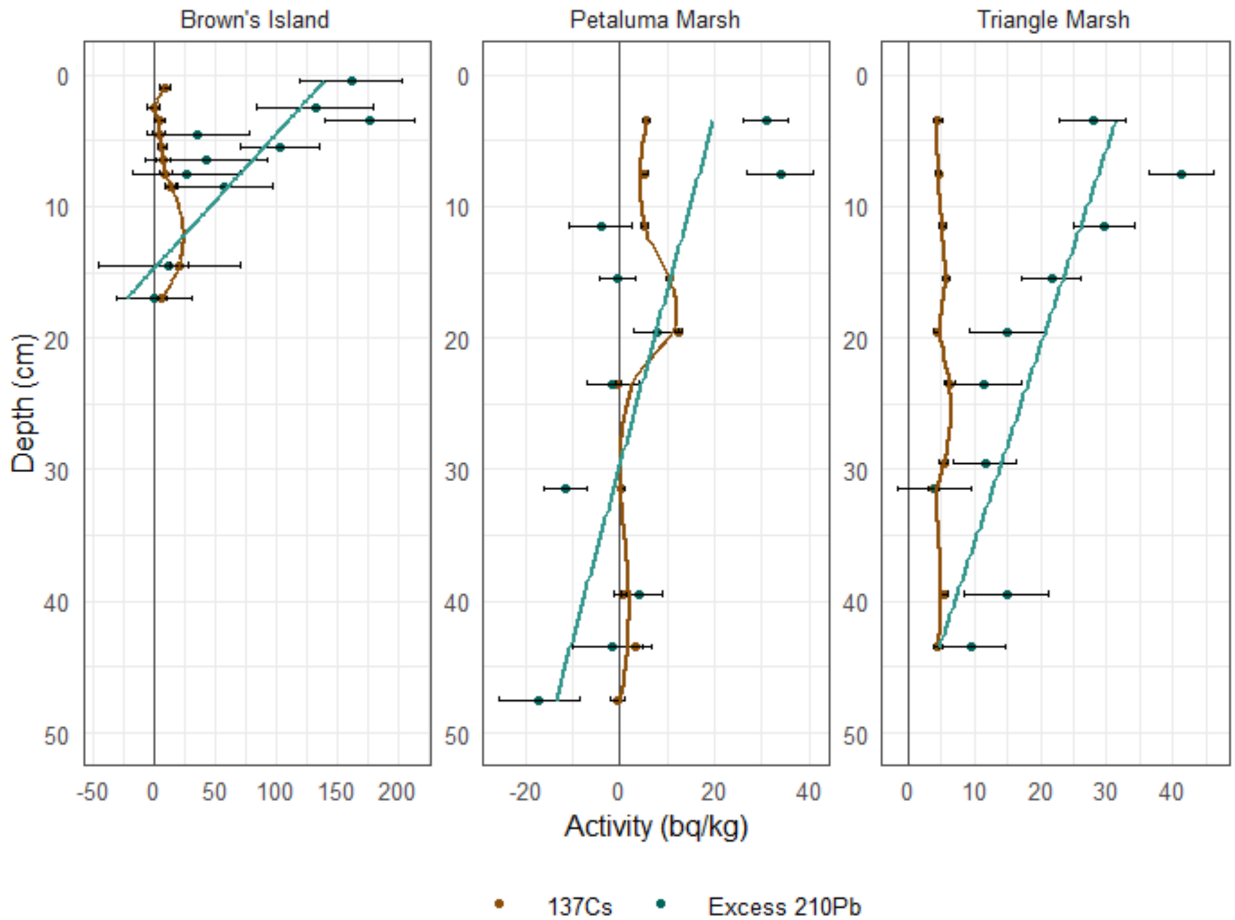
Site Name	^{210}Pb (mm yr^{-1})			^{137}CS (mm yr^{-1})		
	Average	Min	Max	Average	Min	Max
Petaluma Marsh (PTL)	1.7	1.1	3.2	3.8	3.1	4.6
Browns Island (BI)	2.2	1.0	7.0	2.5	1.6	3.3
Triangle Marsh (TRM)	1.9	1.2	3.7	--	--	--

Sample name	UCIAMS Lab #	Depth (cm)	Material	D14C (‰)	±	^{14}C age (BP)	±
PTL15-02	183277	223	Plant	-185.3	1.4	1645	15
PTL15-02	183278	411	Plant	-257.4	2.6	2390	30
PTL15-02	183279	652	Plant	-384.4	1.7	3895	25
PTL16-02	160407	799	Plant	-433.7	1.2	4570	20
PTL16-02	191919	988	Plant	-442.5	0.9	4695	15
PTL16-02	191918	1107	Stick	-15.8	1.3	130	15
PTL16-02	185583	1124	Shell	-520.2	0.9	5900	15
PTL16-02	183280	1170	Plant	40.3	1.7	-310	
PTL16-02	191916	1193	Stick	86.2	1.5	Modern	
BI107	185590	107	Plant	-41.6	1.4	340	15
BI197	185591	197	Plant	-110.2	1.4	940	15
BI-15-01_386	191941	386	Plant	-213.5	1.4	1930	15
BI-15-01_564	191942	564	Plant	-294.4	1	2800	15
BI-15-01_753 .017mgC	191943	753	Plant	-383.5	10.5	3890	140
BI-15-01_950 .15mgC	191944	950	Plant	-392	1.1	4000	15
BI1-1054	168450	1054	Plant	-460.4	0.9	4955	15
BI1-1055 .025mgC	168451	1055	Plant	-461.7	6.3	4980	100
BI1-1061 .28mgC	168452	1061	Plant	-452.1	1	4835	15
TRM16-02-149	210748	149	Plant	-39.9	4.4	325	40
TRM16-02-241	185586	241	Plant	-42.6	1.4	350	15
TRM16-02-325	183281	325	Plant	-103.2	1.5	875	15
TRM16-02-427	185587	427	Plant	-154.3	1.3	1345	15
TRM16-02-511	183282	511	Plant	-211.1	1.3	1905	15
TRM16-02-539	210749	539	Plant	-337.3	8.6	3310	110

Table 2.1 Accretion and radiocarbon dates

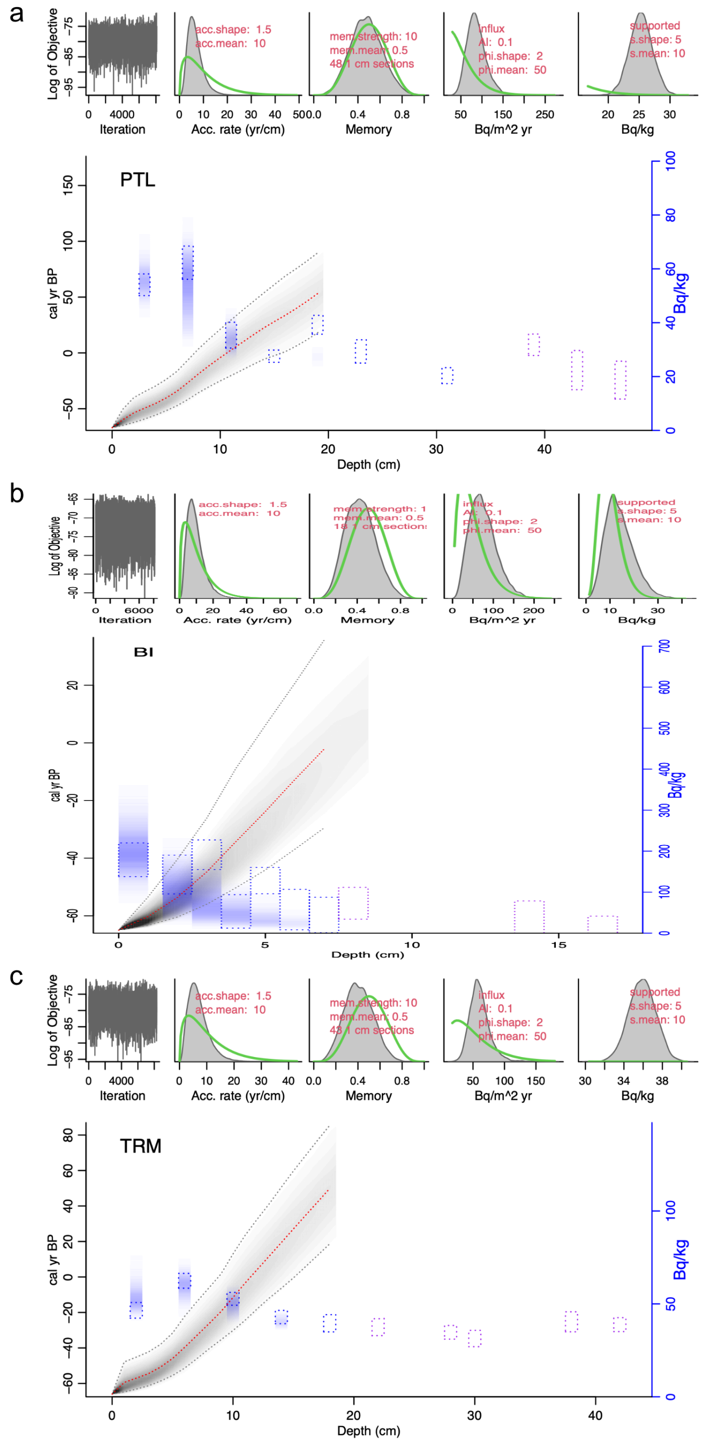
Accretion rates in mm yr^{-1} based on ^{210}Pb and ^{137}Cs, and radiocarbon dates based on ^{14}C, for all sampling sites. Accretion rates obtained from PEARL, Queen's University and USC. Radiocarbon dates obtained from UC Irvine Keck Radiocarbon lab.

2.10 Appendix



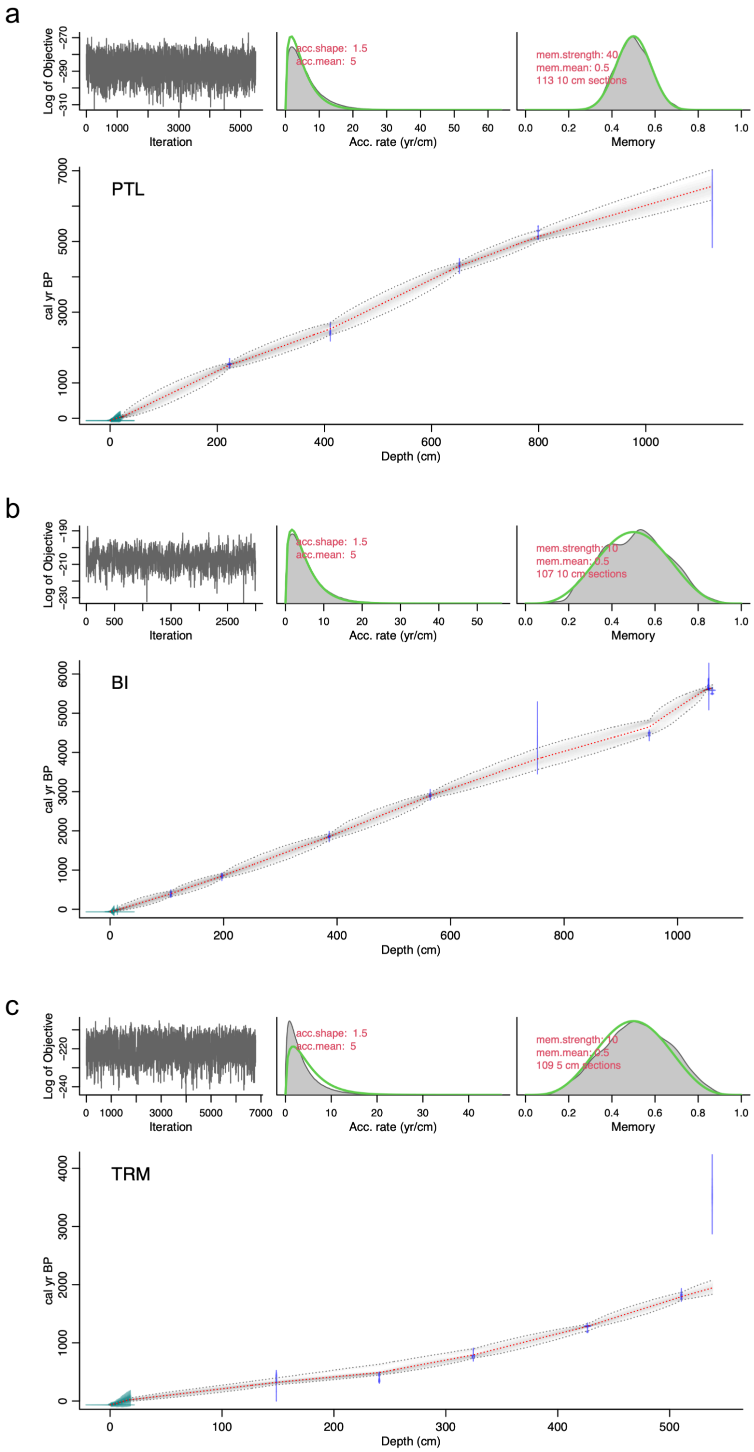
Appendix Figure 2.1 Lead and Cesium core profiles

Core Profiles for ^{137}Cs, Total ^{210}Pb, and Unsupported ^{210}Pb Activities



Appendix Figure 2.2 Plum age-depth models

^{210}Pb Age-Depth Model Output for Plum for a) Petaluma Marsh, b) Browns Island Marsh, and c) Triangle Marsh.



Appendix Figure 2.3 Bacon age-depth models

Bacon Age-Depth Models for a) Petaluma Marsh, b) Browns Island Marsh, and c) Triangle Marsh.

Site	Core year	Upper Depth	Lower Depth	Total ^{210}Pb (bq/kg)	^{210}Pb Unsupported (bq/kg)	^{137}Cs (bq/kg)
Brown's Island	2015	0	1	178.9 ± 41.1	161.7 ± 41.6	9.1 ± 4.5
Brown's Island	2015	2	3	142.9 ± 47.3	132.5 ± 48	0 ± 5.2
Brown's Island	2015	3	4	191.2 ± 36.1	177.1 ± 36.6	4.5 ± 4
Brown's Island	2015	4	5	53.1 ± 40.9	36.1 ± 41.5	4.6 ± 5.1
Brown's Island	2015	5	6	128.3 ± 32.1	103.7 ± 32.5	6.6 ± 3.7
Brown's Island	2015	6	7	57.5 ± 49.2	42.8 ± 50	7.5 ± 5.9
Brown's Island	2015	7	8	44.2 ± 43.3	26.7 ± 44	9.3 ± 5.2
Brown's Island	2015	8	9	72.8 ± 38.8	57.9 ± 39.4	14 ± 4.7
Brown's Island	2015	14	15	22.1 ± 56.5	12.8 ± 57.5	20.7 ± 7.1
Brown's Island	2015	16.5	17.5	11.5 ± 30.1	0 ± 30.6	6.3 ± 3.8
Petaluma Marsh	2015	3	4	54.1 ± 4	30.9 ± 4.9	5.6 ± 0.7
Petaluma Marsh	2015	7	8	62.3 ± 6.2	34 ± 7.1	5 ± 0.9
Petaluma Marsh	2015	11	12	35.4 ± 4.8	-4.1 ± 6.6	5.2 ± 0.8
Petaluma Marsh	2015	15	16	27.6 ± 2.3	-0.5 ± 3.8	10.5 ± 0.6
Petaluma Marsh	2015	19	20	39.2 ± 3.5	7.7 ± 4.9	12.6 ± 0.8
Petaluma Marsh	2015	23	24	29.3 ± 4.3	-1.5 ± 5.6	-0.5 ± 0.6
Petaluma Marsh	2015	31	32	20.3 ± 3	-11.5 ± 4.6	0.3 ± 0.6
Petaluma Marsh	2015	39	40	31.8 ± 4	3.9 ± 5.2	0.7 ± 0.6
Petaluma Marsh	2015	43	44	22.4 ± 7.3	-1.6 ± 8.3	3.4 ± 1.5
Petaluma Marsh	2015	47	48	18.7 ± 7	-17.1 ± 8.6	-0.6 ± 1.6
Triangle Marsh	2016	3	4	46.5 ± 4.2	27.9 ± 5	4.4 ± 0.6
Triangle Marsh	2016	7	8	62.4 ± 4	41.2 ± 4.9	4.5 ± 0.5
Triangle Marsh	2016	11	12	52.7 ± 3.5	29.6 ± 4.5	5.1 ± 0.5
Triangle Marsh	2016	15	16	42.9 ± 3.6	21.7 ± 4.5	5.6 ± 0.6
Triangle Marsh	2016	19	20	39.6 ± 4.6	15 ± 5.7	4.3 ± 0.6
Triangle Marsh	2016	23	24	37.5 ± 4.5	11.4 ± 5.7	6.2 ± 0.8
Triangle Marsh	2016	29	30	34.7 ± 3.7	11.6 ± 4.7	5.3 ± 0.6
Triangle Marsh	2016	31	32	31.4 ± 4.4	4 ± 5.6	3.8 ± 0.8
Triangle Marsh	2016	39	40	40.4 ± 5.4	14.9 ± 6.4	5.3 ± 0.8
Triangle Marsh	2016	43	44	38.8 ± 3.7	9.6 ± 5.1	4.4 ± 0.6

Appendix Table 2.1 Raw ^{210}Pb and ^{137}Cs data

Raw ^{210}Pb and ^{137}Cs data for Brown's Island, Petaluma Marsh, and Triangle Marsh.

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3. Increasing Salt Marsh Elevation Using Sediment Augmentation: Critical Insights from Surface Sediments and Sediment Cores

3.1 Abstract

Sea-level rise is particularly concerning for tidal wetlands that reside within an area with steep topography or are constrained by human development and alteration of sedimentation. Sediment augmentation to increase wetland elevations has been considered as a potential strategy for such areas to prevent wetland loss over the coming decades. Here, we analyze sediment cores from the Seal Beach National Wildlife Refuge (SBNWR) to determine the nature of the pre-existing sediments of the site and understand natural accretion rates and variability over time. These results are compared to the augmentation sediments and depth of sediment applied during a 2016 experimental augmentation treatment. Although the cores revealed natural variations in the grain size and organic content of sediments deposited at the site over the past 1500 years, the applied sediments were markedly coarser in grain size than prehistoric sediments at the site (100% maximum sand versus 76% maximum sand). The rate of the experimental sediment application (25.1±1.09 cm in ~2 months) was also much more rapid than natural accretion rates measured for the site. In contrast, post-augmentation sediment accretion rates on the augmentation site have been markedly slower than pre-augmentation rates or current rates on the nearby control site. The mismatch between the characteristics of the applied sediment and depth of sediment applied and the natural conditions of the marsh are likely strong contributors to the slow recovery of marsh vegetation observed at the site in the five years following the augmentation experiment. Sediment augmentation has been shown to be a useful strategy in some marshes, but, as the Seal Beach case study illustrates, such projects clearly require careful

regard for naturally occurring sediment characteristics, depositional dynamics and time that may be required for vegetative recovery.

3.2 Introduction

Climate change presents increasing challenges for the management of coastal wetlands. These settings are considered one of the most heavily threatened natural systems globally (Barbier et al., 2019) and are disappearing at unprecedented rates (Finlayson et al., 2019; Thorne et al., 2018). One of the biggest threats to coastal wetlands in the 21st century is sea-level rise (SLR) (Schuerch et al., 2018). Not only can SLR exacerbate short-term flooding events, but it can also increase the recurrence of inundation periods which may exceed the natural thresholds of coastal ecosystems (Bilskie et al., 2014; Voss et al., 2013). Sea levels are very likely to continue rising over the 21st century, affecting 70 percent of coastlines worldwide (IPCC 2022). Estimates for future SLR rates range anywhere from 29 cm to 1 m by the end of the century, depending on greenhouse gas emission rates (DeConto and Pollard 2016). Global projections anticipate between 20 and 90 percent of coastal wetland loss under low and high sea-level rise scenarios, respectively (Schuerch et al., 2018). This threat is exacerbated by the expansion of coastal development, which leaves many coastal marshes bordered by urban and agricultural usage, preventing marsh migration into adjacent uplands, also known as coastal squeeze (Torio and Chmura 2013). Marsh loss due to SLR, coupled with the lack of upland migration area, will greatly reduce the ability of marshes to maintain their biodiversity, as well as areas of refuge for endemic or endangered wildlife species.

Marsh formation and development results from complex interactions between geologic, hydrologic, and ecologic factors that are highly dependent on inorganic sedimentation supply from estuarine sources, especially in areas with little upland inputs (Schile et al., 2014; Byrd and Kelly 2006). In order to develop and maintain optimal elevation levels, salt marshes must have protection from high-energy waves that would erode soil otherwise used for accretion, while also providing source materials (mainly silt, clay, organic matter, infrequently fine sand) from low-wave energy tides (Davidson-Arnott et al., 2002). This sensitivity allows for marshes to respond rapidly to fluctuations in sea level by adjusting their rates of accretion (Adam 1990; French 2006).

Research also shows that microtidal coastal marshes can accrete vertically and expand horizontally quite rapidly as a result of storms, including hurricanes (Craft et al., 1993, Schuerch et al., 2012, Thorne et al., 2022). As such, these ecosystems can be reliant on storms to supply sediment, which is useful for quick platform development but may not be sustainable in keeping pace with accelerating SLR (Townend et al., 2011) depending on the marsh location. However, salt marshes in Mediterranean climates with less severe and infrequent storms cannot rely on sediment supply from storms due to the inconsistent nature of these events. In addition, vegetation plays an important role as salt marsh plants vegetate the marsh platform and aid in marsh accretion through particle capture of fine-grained sediment and aggradation of accumulated vegetation debris, enhancing overall sediment deposition potential (Morris et al., 2002, Leonard and Croft 2006, Perillo et al., 2019). However, it is important to note that while sedimentation is necessary for a salt marsh's ability to survive and thrive, an excessive rate of mud deposition can damage the existing vegetation and diminish overall ecological function (Bird 2011; Stagg and Mendelssohn 2010).

One approach to address coastal erosion and build marsh elevation relative to sea level has been through sediment addition (Pope 1997; Berkowitz et al., 2017; Ganju 2019). Some approaches place dredged sediments in the nearshore zone with the intent of subsequent tidal or storm redistribution (Schwartz and Musialowski 1980; Fettweis et al., 2011). Sediment addition projects can also focus on augmenting marsh sediment cover by adding dredge materials directly on top of areas of eroding or scouring marsh surfaces to increase marsh elevations relative to sea level. Previous sediment augmentation efforts have been conducted in a number of settings including Essex, UK (Widdows et al., 2006); Venice, Italy (Scarton and Montanari 2015); Narrow River Estuary, RI (Wigand et al., 2017); along the Mississippi River delta region in southern Louisiana (La Peyre et al., 2009; McCall and Greaves 2022); and in various US states including New Jersey (VanZomeren et al., 2018), New England (Perry et al., 2020; Puchkoff and Lawrence 2022), North Carolina (Davis et al., 2022), and California (Thomsen et al., 2022). Such efforts have had mixed results. Widdows et al. (2006) found that, following the placement of fine dredge material (ca. 0.6 m depth) on the upper shore at 2 estuaries situated in Essex, UK, short-term erodibility was high, but long-term temporal changes in sediment erodability reflected the nature of benthic assemblages established during the recovery period (19 months). La Peyre et al. (2009) found that, following sediment addition at six brackish marsh sites located in the Mississippi River delta region in southern Louisiana, vegetative cover and productivity response were minimal for deteriorating vegetated marshes, with short-term data showing no significant impact of sediment enhancement and long-term trends indicating decreasing productivity over time. While sediment addition is not a new approach, the mechanical and fiscal constraints of these large-scale projects have limited the number of examples. Furthermore, the regional variability of the environments requires more examples to provide a better understanding of how to implement and support future sediment augmentation efforts and make them more successful in combating loss due to SLR.

A sediment augmentation project at Seal Beach National Wildlife Refuge (NWR), southern California, USA provides the first opportunity in this vulnerable coastal region (Thorne et al., 2018) to test the effect of thin-layer sediment application to salt marsh surfaces, with the goal of mitigating habitat loss caused by accelerated SLR. This is through provision of additional mineral material for elevation gain, with the aim to be able to use this strategy at regular intervals for long-term sustainability of marshes despite SLR.

In this study, we examine the behavior of augmentation sediments, as well as past sediments and their depositional dynamics at the Seal Beach augmentation site through the use of sediment cores. We examine whether the material used in the augmentation project is similar to sediment found in the current and prehistoric natural environment. We also seek to understand natural accretion and variability over time, and how it compares to the artificial accretion from sediment addition. Specifically, we ask 1) What is the grain size and thickness variability of the sediment applied to the newly augmented salt marsh platform? 2) How does augmentation sediment grain size compare to recent and prehistoric sediment at the augmentation and control site? and 3) How different is the accretion rate of the augmentation layer compared to the natural accretion seen historically in the environment?

3.3 Methods

3.3.1 Study site

This study was conducted at the Seal Beach NWR (Figure 3.1), which is managed by the US Department of the Interior, U.S. Fish and Wildlife Service (USFWS). The refuge is located in Orange County, California, USA within the Naval Weapons Station Seal Beach (33˚ 44' 17.99" N, -118˚ 03' 60.00" W), and spans 965 acres, with 750 acres of tidal marsh, including three intertidal and subtidal restored ponds (McAtee et al., 2020). The refuge consists of approximately 390 hectares of relatively undisturbed salt marshes and is the only remaining undeveloped part of the Anaheim Bay estuary, although surrounded by reclaimed areas of military, municipal, and industrial infrastructure. The climatic and oceanographic settings at Seal Beach NWR are typical of Southern California, with hot/dry summers and mild winters, and semidiurnal tides with a mean micro-tidal range of <2m (Avnaim-Katav et al., 2017). The marsh harbors federally endangered species including the light-footed Ridgway's rail (*Rallus longirostris levipes*), the California least tern (*Sternula antillarum browni*), and the Belding's savannah sparrow (*Passerculus sandwichensis beldingi*). Pacific cordgrass (*Spartina foliosa*) and pickleweed (*Salicornia pacifica*) dominate the vegetative landscape, with cordgrass representing 260 ha of the salt marsh platform (Thorne et al., 2019).

Historically, the refuge wetlands received sediment input from episodic storm surges as well as flows from the Santa Ana and San Gabriel Rivers, allowing the refuge to keep pace with SLR in the region (0.98 ± 0.23 mm/yr; NOAA station 9410660) (Grossinger et al., 2011; Rosencranz et al., 2017). Before the twentieth century, the salt marsh at Seal Beach became isolated from the Santa Ana River, due to channelization for flood control, making the refuge more vulnerable to accelerated SLR due to a lack of terrestrial sediment input (Leeper 2015; Kirwan and Megonigal 2013). Additionally, 4.13 mm yr^{-1} of subsidence has been observed in the region, likely due to oil, gas and water extraction between 1994 to 2012 (Takekawa et al., 2014). The refuge, which is situated along the San Andreas Fault, has also suffered elevation loss due to tectonic subsidence (Leeper 2015). These compounding alterations to the system and subsidence, coupled with increased SLR in the region, make Seal Beach NWR especially vulnerable, with one study estimating that the rate of relative SLR is three times higher than that of nearby marshes in the southern California region (Takekawa et al., 2013).

3.3.2 Sediment augmentation

3.3.2.1 Construction background

The USFWS designed and implemented the sediment application methodology, along with the help from university, state and federal partners. The Environmental Management Agency, later known as Orange County Public Works (OCPW), managed the dredging, construction of sediment barriers, and sediment application of the project.

The monitoring timeline spanned 6 months prior to the sediment augmentation addition to 5 years post-augmentation. The goal was to uniformly place 13,500 cubic yards of dredge material, thinly spread over 4.05 ha, to achieve a 25 cm (10”) of sediment depth and to maintain a minimum of 7.6 cm (3”) increase on the marsh platform 2 years after sediment addition (McAtee et al., 2020). The sediment material was sourced from an adjacent subtidal area near the refuge, within the Anaheim Bay. Sediment materials from the dredge site were tested for chemical contaminants and grain size compatibility, along with sediment materials from the proposed augmentation site. The proposed dredge materials were deemed to be clean and compatible when compared to the augmentation site materials by Orange County Parks and USFWS (Sloane et al., 2021).

3.3.2.2 Sediment addition

A total of 12,901 cubic meters (16,874 cubic yards) of dredge material was placed, with an average depth of 22 cm across the site (Thorne et al., 2019). The sediment material was applied in stages, using sediment spray equipment, with the first application occurring between January 22, 2016, and April 4, 2016. A variety of mitigation measures were taken, including relocating rail nesting platforms, maintaining a 50 ft. vegetated buffer zone from the water’s edge, silt barriers around the augmentation site, in-water silt curtains for dredge operations, and maintaining bio-monitors on site (USFWS, pers. comm. Rick Nye). Engineering interventions such as hay bales, straw waddles, sandbags, and geotextile fabrics were placed along the perimeter of the augmentation site to retain the sediment material (Thorne et al., 2019). Dredging challenges arose when obtaining the sediment augmentation material, which appear to have resulted in sandier grain sizes and lower organic matter compared to the original topsoil at the augmentation site (McAtee et al., 2020).

3.3.3 Surface sediment samples

3.3.3.1 Grain size sampling methods and laboratory techniques

Following augmentation, 113 surface sediment samples were collected and used in this study (Figure 3.1). Samples were opportunistically collected immediately after sediment application by USFWS employees (R. Nye, K. Gilligan). Grain size was analyzed for these samples using the Bouyoucos hydrometer method (Bouyoucos, 1962), and hydrometer readings and temperatures were recorded immediately (to determine the % sand) and two hours later (to determine the % silt and % clay).

3.3.3.1.1 Kriging-based spatial interpolation with grain size

The one hundred and thirteen surface grain size samples were analyzed using the (Sibson) *kriging* interpolation method (Figure 3.2). Kriging has been widely used as a geostatistical method in soil science to explore surface variations using spatial correlation methods along a spatially correlated distance (Zhang et al., 2020; Liu et al., 2006; Gotway et al., 1996; Sibson 1980). A total of three maps were created using the Natural Neighbor tool in

ArcGIS to visualize the spatial variability of clay, silt and sand values associated with each surface sample taken along the augmentation site following the surface sediment application.

3.3.3.2 Sediment thickness

Measurements of the sediment augmentation thickness were distributed across the entire area with the expectation that the sediment addition would not be uniform, and with the goal to provide representative sampling across the entire area of sediment addition (excluding the buffer area). Although the construction target was even distribution of augmented sediment across the entire project area, spatial heterogeneity was expected. Thus, sediment thickness was sampled at multiple locations across the project area.

Sediment stake stations were established in the augmentation Site. The sediment stake stations were located on a 20 m grid across the entire sediment augmentation area to ensure even coverage of the site; wide distribution of sediment stakes provides a good assessment of spatially variable sediment thicknesses. Seventy-one stakes were measured during sampling. Some stakes from the original grid were missing after sediment addition, either because they were inadvertently removed during the sediment addition or because so much sediment was added that the tops of the stakes were buried. The purpose of the sediment stake grid was to provide a more comprehensive spatial assessment of sediment thickness. Because no sediment was added to the control area, a sediment stake grid was not established.

3.3.3.2.1 Survey timing and field methods

Post-augmentation sampling began in June 2016, two months after the completion of sediment addition. For the first two years, sampling occurred about every six months. The next sampling occurred 12 months later, in 2019, three years after sediment addition. The final sampling occurred in June and July 2021, 62 months after the sediment was added.

Sediment stakes (¾" Schedule 80 PVC pipes) were placed in the sediment with a known height (55 cm) above the substrate. Sediment stakes are commonly used in sediment accretion studies and the protocol is well developed (Roegner et al., 2008). Sediment accretion (accumulation) or erosion was determined by measuring the distance from the substrate surface to the top of the stake. Since all stakes were installed with precisely 55 cm between the ground surface at time of installation and the top of the stake, the thickness of the added sediment was determined as the difference between 55 cm and the measured distance at time of sampling. This length was chosen to ensure that approximately 30 cm (11.8") would be exposed after the sediment was added to a depth of about 25 cm (10"). Having only 30 cm exposed after sediment augmentation reduced the possibility of predatory birds using the sediment stakes as perching locations.

3.3.3.2.2 Kriging-based spatial interpolation with sediment thickness

The fifty-five measured sediment thickness datapoints were analyzed using the (Sibson) *kriging* interpolation method (Figure 3.3). A total of two maps were created using the Natural Neighbor tool and Contour tool in the 3D analyst box in ArcGIS to visualize the five interval classifications for sediment thickness (0-6, 6-15, 15-23, 23-27, 27-35, and 35-60 cm).

3.3.4 Sediment cores

3.3.4.1 Sampling site locations and field procedures

Sediment cores were obtained prior to augmentation using a Russian Peat Borer, which takes 1 m lengths of 2.5 cm diameter sediment cores while minimizing compression of sediment samples. Sites were selected in the field with an effort to obtain good geographic coverage and variation in extant plant coverage (pre-augmentation conditions) on both the control and augmentation sites, while maximizing distance from marsh channels which might have impacted the long-term records due to meandering (Figure 3.1). To ensure adequate sampling coverage, material, and replicability, three cores were taken on the control site and three cores were taken from the augmentation site. All cores collected vary from 1 m to 2 m in total. At each core location a GPS point was taken, and vegetation of the surrounding area was described. All samples were extruded in-field, described, and wrapped in plastic wrap for transport back to University of California, Los Angeles (UCLA), where they were stored at in a cold room at 4°C.

3.3.4.2 Initial core analysis

Within 10 days of collection, sediment cores underwent initial description and analysis. Cores extruded in the field were unwrapped, photographed, re-measured for any shrinkage or expansion, and visually described. Following these preliminary analyses, cores were split in half down to 50 cm depth. One half of the top 50 cm of each core was sent to CSULB for analysis of below ground biomass, while the remaining half was analyzed at UCLA for radiometric activity and carbon content.

3.3.4.3 Chronological control

3.3.4.3.1 Radiocesium and radiolead preparation

For chronological control over the past century, ^{137}Cs and ^{210}Pb have been used to determine recent sedimentation (Zhang et al., 2015). These isotopes were used for all six of the cores, and ^{14}C dating was used for five of the cores to provide an age-depth model. Based on previous measurements of the ^{137}Cs bomb spike depth (1961-63) in Seal Beach sediments, accumulation rates in the area ranged from 2.2 – 4.6 mm yr^{-1}. Consequently, cores were sectioned in 2-4 cm intervals, to a minimum of 20 cm (for low-accreting sites in the high marsh) and a maximum of 60 cm depth (for high-accreting sites in the low marsh). After sectioning, samples were dehydrated in a drying oven at 110°C for 24 hours and then weighed to calculate bulk density (g/cm^3). Samples were lightly ground, sealed in plastic tubes (1 cm OD, sample heights 2-3 cm), reweighed, and sent to the University of Southern California (USC) for ^{137}Cs and ^{210}Pb analysis.

3.3.4.3.2 Excess radiocesium and radiolead

Excess ^{210}Pb ($^{210}Pb_{ex}$) and ^{137}Cs activities in sediments were measured using high-purity intrinsic germanium well-type detectors (ORTEC, 120 cm^3 active volume). Detector efficiencies were determined by counting standards in a similar geometry. Standards used included IAEA-385 marine sediments, EPA diluted pitchblende (SRM-1), and NIST ^{210}Pb liquid solution (SRM 4337). Samples were counted for 2–4 days, to measure the following: ^{210}Pb, ^{214}Pb, ^{214}Bi, and ^{137}Cs. Standards were 3.0 cm high, and corrections were made to account for the different sample heights used. The ^{226}Ra activity (supported ^{210}Pb) was determined from the ^{222}Rn daughters (^{214}Pb and ^{214}Bi). A small (10 %) correction was applied to each sample to account for radon leakage, based on measurements of radon loss from similar sediments. Excess ^{210}Pb was determined by subtracting the supported ^{210}Pb from total ^{210}Pb and correcting for decay between collection and analysis.

Two models can be applied to determine sedimentation rates from $^{210}Pb_{ex}$ profiles: the constant rate of supply (CRS) model and the constant initial concentration (CIC) model. Both models assume a time-independent flux of ^{210}Pb across the sediment water interface (SWI) and the CIC model also assumes that sedimentation rates are time-independent (Benninger and Krishnaswami, 1981; Robbins and Edington, 1975; Robbins, 1978; Appleby, 2001; Kirchner, 2011). For the CRC model, excess $^{210}Pb_{ex}$ inventories were calculated by multiplying excess activity by bulk density and integrating the result downcore. For unmeasured intervals, assumptions were made. Sediments above the top section measured were assumed equal to those in the top interval measured. Linear interpolations were made for deeper gaps. When $^{210}Pb_{ex}$ appeared to be zero for consecutive intervals, the integration was terminated. Error propagation was applied to evaluate uncertainties for missing intervals. Errors for ages determined by the CRC model were calculated by a Monte Carlo approach. Briefly, 1000 random values were generated for each depth interval based on $^{210}Pb_{ex}$ uncertainties for that interval. After the 1000 $^{210}Pb_{ex}$ values were used to determine the interval age, the 1000 ages were averaged, and its standard deviation was calculated. Uncertainties are modest near the top of the core but become quite large as ages reach 2-3 ^{210}Pb half-lives. The CIC model gave comparable accumulation rates for each core.

^{137}Cs concentrations were often low but gave an indication of the 1961-63 peak from atmospheric weapons testing. A depth range for the age of this horizon was estimated by selecting the observed maximum for the ^{137}Cs profile and assuming the actual maximum was midway between this horizon and the subsequent interval.

3.3.4.3.3 Radiocarbon

For ^{14}C dating, organic macrofossil samples for ^{14}C were visually identified, extracted from the core, rinsed with DI water, dehydrated in a drying oven at 110°C for a minimum of 1 hour, weighed, wrapped in plastic, and taken to the UC Irvine Keck Radiocarbon Lab for final processing. A total of eight plant macrofossil samples were dated (see Appendix Table 3.1). Because any root matter will introduce erroneously young ^{14}C ages into older sediments, all plant-matter was identified as above-ground leaves or seeds. Radiocarbon dating was conducted using a 500 kV compact AMS (accelerator mass spectrometer) unit from National Electrostatics Corporation. Plant macrofossil samples and carbonate samples were pretreated following

KCCAMS/UCI facilities hydrogen reduction method (Santos et al., 2007). Plant macrofossil organic materials were calibrated using IntCal20 terrestrial calibration curve (Reimer et al., 2020). Age estimates and uncertainties for all ^{210}Pb, ^{137}Cs, and ^{14}C ages were incorporated into a single Bayesian age-depth model using the package rbacon version 2.5.3 with IntCal version 0.1.3 in the R interface (Blaauw and Christen, 2013, Rstudio Team, 2020). All ^{14}C ages are reported with 1950 CE as "Present".

3.3.4.4 Sedimentological analysis

In this study, loss-on-ignition (LOI) was completed for all cores to a depth of 100 cm. Bulk density was also identified, defined as the mass of organic and mineral components, divided by a wet volume of 1 cubic centimeter (Morris et al., 2016). Sediment cores were sliced into 1 cm intervals. From each slice, a 1 cubic centimeter sample was extracted, dehydrated in an oven overnight, burned at 550 ºC for 4 h, and at 950 ºC for 1 h to measure the water content as a percentage of wet weight, bulk density in grams per cubic centimeter, organic content as a percentage of bulk density, and carbonate content as a percentage of bulk density, following standard protocols from Heiri et al. (2001). Remaining material is interpreted to be non-carbonate inorganic sediment component.

3.3.4.5 Below ground biomass

Below-ground responses of marshes to environmental factors, such as sea-level rise, have been found to be more broadly applicable than above-ground feedbacks due to consistency between plants and a lack of dependability on mineral sediment availability (Kirwan and Guntenspergen 2012). Below-ground root biomass, in particular, has been found as an indicator of plant health in marsh environments when compared to aboveground biomass (Turner et al., 2004). The top 50 cm of each sediment core was used to calculate belowground biomass, with the exception of core SB15_06, which is missing the top 20 cm of sediment. Sediment cores were sieved in 4.75 mm sieves, and small plant roots were rinsed, bathed in fresh water, and dried to remove soil and debris. Dried sieved plant matter (bulk, not separated by type) was then submerged in water in a graduated cylinder to record the volume. Plant roots were then dried, wrapped in pockets of foil, labeled, and placed in a drying oven for 24 hours. After drying for at least 24 h at 100-110˚C, the roots were weighed. This measurement is below ground biomass per unit area (surface area cored).

3.3.4.6 Grain size analysis

For the three sediment cores from the augmentation site and the three sediment cores from the control site, the sampling strategy for grain size aimed to maximize the temporal resolution in the top 1 m (approximately 100-300 YBP). Above 1 m depth, a sample was taken every 2 cm; below 1 m depth, samples were extracted every 5 cm. A total of 255 samples in total were successfully analyzed.

Samples were approximately 0.5 cm^3 when extracted. They were boiled with 25-30 mL of 30% H_2O_2, until reactivity ceased, indicating full removal of organic particles. Samples were then transferred to vials which were transported to California State University Fullerton to the Paleoclimatology and Paleotsunami Laboratory, where they were analyzed using a Malvern Mastersizer 2000 Laser Diffraction Particle Size Analyzer coupled to a Hydro 2000G large-volume sample dispersion unit. Laboratory procedures are further explained in Kirby et al. (2015). Particle sizes were classified as sand, silt, or clay based on the Wentworth scale.

Results were plotted using Bayesian age-depth models obtained from R software Bacon (Blaauw 2010) where possible. For those sections of core which were analyzed for grain size, but were below the lowest ^{14}C date obtained (or were from a core not ^{14}C dated, as in the case of samples from SB15_21), a linear age-depth model was extrapolated by obtaining the average sediment accumulation rate over the Bayesian model (2.1 mm yr^{-1} for SB15_09; 1.9 mm yr^{-1} for SB15_11; 1.7 mm yr^{-1} for SB15_20) or using the ^{137}Cs-obtained accretion rate (2.5 mm yr^{-1} for SB15_21). For sediment cores with an age-depth model, the last modeled age was used to start the linear extrapolation.

3.3.4.7 Net sediment accretion rates

Sediment accretion was measured using two methods: feldspar plots and radioisotope analyses of sediment cores. Feldspar plots were created with PVC stakes marking the corner of the plots in the augmentation and control sites before the augmentation sediment layer was added. Feldspar provides a white marker horizon representing the marsh surface before sediment accretion (Cahoon and Turner 1989). To measure sediment accretion rates after sediment addition, additional feldspar plots were established on top of the added sediment by sprinkling (when the plot was exposed to air) 1200-1600 mL of dry Custer Feldspar clay within the perimeter of a 0.5 m by 0.5 m quadrat. The thickness of sediment accumulated on top of the plots was measured by taking a triangular wedge-shaped “core” using a knife, and measuring the thickness from the top of the feldspar layer to the top of the sediment; three measurements were taken, one on each side of the triangle, and averaged. If feldspar was visible on the surface of the plot, the thickness was recorded as zero.

Cesium and lead measurements were taken from the sediment cores pre-augmentation. Net marsh sediment accretion rates for the modern period are based on total depth of marsh sediment accumulated at each core following the 1963 ^{137}Cs peak. Longer-term marsh sediment accretion (> 60 years) is based upon the total depth of marsh sediment accumulated in each core with the initiation of marsh sedimentation determined by ^{210}Pb or ^{14}C dating. Depths are divided by time to derive total sediment accretion rates (Figure 3.7).

$$\textit{Net Marsh Sediment Accretion Rate (mm yr}^{-1}\textit{) = Depth Marsh Sediment (mm)/Time (yr)}$$

3.3.5 Permutational Multivariate Analysis of Variance

Permutational Multivariate Analysis of Variance (PERMANOVA) is a non-parametric multivariate statistical test, which does not rely on the assumptions of normality and equal variances. In this study, PERMANOVA was run on the grain size samples to compare a) the top 10 cm of each core, b) all the surface grain size samples, and c) the bottom portion of three cores of which sand represents <20% (SB15_09, SB15_11, SB15_20) compared to the surface grain size samples, to deduce differences between grain size of all the sediments (Table 3.1) (Anderson 2014; Anderson 2001). The permutational analysis was performed based on the Euclidean Distance similarity matrix. Permutational Analyses of Multivariate Dispersions (PERMDISP) was tested in conjunction with PERMANOVA to identify location vs. dispersion effects, and to look for differences between levels within factors (Anderson and Walsh 2013). All the statistical tests and figures were performed in RStudio (2020).

3.4 Results

3.4.1 Surface sediment samples

Grain size

A total of 113 surface grain size samples from the post-augmentation surface were used in this study (Figure 3.1). Spatial distribution patterns of grain size variability of clay, silt, and sand in the augmentation sediment layer are illustrated in Figure 3.2. Light green to dark green on the maps represents concentration levels in percent units. Once distributions of silt and clay are identified and distributed, the remaining material is presented as sand. Sediment clay, silt, and sand contents from the augmentation sediment layer ranged from 0.0 – 51.6%, 0.0 – 58.0%, and 2.1 – 100.0%, respectively. Due to how samples were consolidated during initial testing of the source material, the amount of sand at the selected dredge locations was under-estimated. The grain size of the dredge material contained much less silt and clay (15%) than the pre-sediment application grain size or the control site (57%, 38% respectively) (McAtee et al., 2020). Unlike the original marsh sediment grain size, the applied sediment and the sediment on the experimental site after sediment application was low in silt and clay content (16%) (Figure 3.2).

The highest percent clay content was located in a small segment at the northern region of the site, as well as throughout a larger segment concentrated along the southern portion of the sediment (Figure 3.2a) Similarly for percent silt content, the highest concentrations are found along the southern portion of the sediment, as well as scattered throughout the middle to northern portions of the sediment (Figure 3.2b). The largest dissimilarities can be found in comparison with the percent sand content. We also see the contrast between regions with highest clay and silt percent concentrations, indicated with lighter green shading, and regions with highest percent sand content, indicated by dark green shading (Figure 3.3c). The augmentation sediment layer clearly had higher concentrations of sand through the majority of the site when compared to clay and silt concentrations.

Sediment thickness

Two months after sediment was added to the augmentation site, the added sediment had a depth of 25.1±1.1 cm (Mean±SE). This is essentially equal to the target depth of 25 cm. Mean depths varied over time with no clear trend. At 62 months, sediment depth was 23.9±1.2 cm. The median was lower than the mean for all times, reflecting the influence of a few large values on means. Every sampling period was characterized by a wide variability in depths. Two months after sediment addition, the range was 3.7 to 52.5 cm. The range for successive samples was similar, with a range of 1.7 to 51.9 cm at 62 months.

The spatial variability in sediment depths is illustrated in the depth contour maps of the sediment stake data (Figure 3.3). Two months after sediment addition, there were some distinct areas of thinner and thicker sediment depths. The eastern half of the study site had mostly moderate sediment depths in the 23-35 cm range, although there were a few localized spots with thinner sediment less than 23 cm deep. In contrast, the northwestern quadrant had relatively thin sediment (15-23 cm deep) and the southwestern quadrant had thick sediment (>35 cm deep). The changes in sediment thickness five years after the sediment was added shows that the eastern side of the study site mostly decreased in thickness while the western side mostly increased. Most of the changes were modest, either 0-5 cm decrease or 0-3 cm increase, although there were a few isolated pockets of larger changes. Although some areas experienced moderate changes in sediment depth, the average across the entire site was only a modest decline of about 1 cm from 2016 to 2021.

3.4.2 Sediment cores

3.4.2.1 Chronological control

3.4.2.1.1 ^{137}Cs and ^{210}Pb$_{ex}$

Average ^{137}Cs- and ^{210}Pb$_{ex}$-measured accretion for three cores from the augmentation site were 2.9 ± 0.8 mm yr^{-1} and 3.3 ± 0.8 mm yr^{-1} respectively, with average ^{137}Cs-measurements showing slightly lower accretion rates compared to ^{210}Pb-measurments (Table 3.2). Average

^{137}Cs- and ^{210}Pb-measured accretion for three cores from the control site were 3.9 ± 0.9 mm yr^{-1} and 2.5 ± 0.6 mm yr^{-1} respectively, with average ^{137}Cs-measurements showing slightly higher accretion rates compared to $^{210}Pb_{ex}$-measurments. Variation in accretion rates between control and augmentation for all methods was consistently 0.4 – 1 mm yr^{-1}, with the control site average ~ 0.1 mm yr^{-1} higher than the accretion rate at the augmentation site. The consistency between the sites indicates that these sites are suitable for comparison between vertical accretion as the augmentation study progresses.

3.4.2.1.2 Radiocarbon

The uncalibrated and calibrated results from ^{14}C dating of the six cores appears in Appendix Table 3.1. Radiocarbon results from the eight samples analyzed for the six cores returned a maximum age of 1502 ± 126 YBP for a 2 m core (SB15_20) taken in the augmentation site, while the youngest date returned was a 1 m core (SB15_06) taken from the control site at 380 ± 78 YBP. By taking an average of long-term accretion rates from ^{14}C dates, estimated average sediment accretion at Seal Beach NWR is 1.7 ± 0.25 mm yr^{-1}. Two radiocarbon dates, one for SB15_11 and one for SB15_20, produced anomalously young dates. However, all radiocarbon dates were used to create Bayesian models for all sediment cores which have been ^{14}C dated (see Appendix Figure 3.1).

3.4.2.2 Sedimentological analysis

The stratigraphic columns for the top 1 m of each core shows that the top 10 cm of each core is indicative of a richly vegetated marsh platform for both sites (Figure 3.4). Higher organic marsh peat sections vary, with the augmentation site cores having higher marsh peat segments and the control site cores having more silty peat and silty clay segments throughout the cores.

Bulk density concentrations for all cores (Figure 3.4) steadily declined over time. Peak bulk density concentrations are at a maximum of 1.6 g cm^{-3} at around 1000 cal YBP in core SB15_16, with lowest concentrations of 0.1 g cm^{-3} around 20 years ago in core SB15_21. Carbonate content percent has steadily increased in modern times (post-1950s) in all cores apart from SB15_16, which peaked at 49.4 % carbonate content around 250 cal YBP and has steadily declined since. Interestingly, the lowest carbonate content is found in the same core at 3.1 % around 1000 cal YBP. Similarly, organic content still increased over time for one of the cores from the control site, and the other two cores had high organic content variability at intermediate times (these two cores are also the only cores entirely dominated with marsh peat). For the augmentation cores, more variability is present. Peak organic percent content reaches 21.8 % at around 250 cal YBP in core SB15_09, with lowest concentrations of 1.0 % around 490 years ago in core SB15_11.

3.4.2.3 Belowground biomass

The vertical profiles of below ground dry biomass percent for the top 50 cm of each core can be seen in Figure 3.4. For all cores, the lowest percent concentrations can be found towards the bottom of the cores. For SB15_06, below-ground biomass percent peaks at 1.5 % between around 97 and 130 cal YBP, and lowest concentrations of 0.4 % are between 135 and 163 cal YBP. For SB15_09, below-ground biomass percent peaks at 5.0 % between around 93 and 131 cal YBP, and lowest concentrations of 1.5% are between 170 and 304 cal YBP. For SB15_11, below-ground biomass percent peaks at 4.9 % at the top of the core in the very recent past (between 2015 and 1976), and lowest concentrations of 0.8% are between 175 and 215 cal YBP. For SB15_16, below-ground biomass percent peaks at 4.3 % at the top of the core between 1970 and 78 cal YBP, and lowest concentrations of 1.1% are between 135-180 cal YBP. For SB15_20, below-ground biomass percent peaks at 5.2 % at the top of the core in the very recent past (between 2015 and 1989), and lowest concentrations of 1.1% are between 276 and 301 cal YBP. For SB15_21, below-ground biomass percent peaks at 5.3 % at the top of the core between around 111 and 121 linearly extrapolated cal YBP (24 -25 cm depth), and lowest concentrations of 1.2% are between 251 and 261 linearly extrapolated cal YBP (38-39 cm depth).

3.4.2.4 Grain size analysis

Results show that pre-augmentation grain size as represented by the top 5 cm of the cores averages 11% clay, 77% silt, and 10% sand (Figure 3.5). Grain size variability is fairly consistent across cores and between the augmentation and control sites. The maximum sand percentage increases down-core and the highest measured in any sample analyzed was 76%, in core SB15_20. Of the six cores analyzed, three cores (SB15_09 (135-200 cm), SB15_11 (125-180 cm), SB15_20 (115-220)) show periods of high sand concentration (>20%) below 1450 AD where habitat may or may not have been salt marsh as it is today.

The above results compare to post-augmentation grain size measurement taken from February - June 2016, which averaged 9% clay, 10% silt, and 83% sand. Although clay concentrations remained relatively similar in the pre-augmentation and post-augmentation sediment materials, the sand concentration found at the site post augmentation greatly exceeds sand concentrations at the top of the cores in both the control and augmentation sites (pre-augmentation), as well as any sand concentration obtained in analysis of all cores covering a history of 1500 years of accretion.

By plotting the grain size results by age (Figure 3.5), we can estimate that the lenses seen in cores SB15_09, SB15_11, and SB15_20 are an event previously identified as an abrupt subsidence event due to a tectonic event caused by the nearby Newport-Inglewood/Rose Canyon fault system (Leeper et al., 2017). Leeper et al. identify this event as having occurred from approximately 1320 AD to 1590 AD. This matches the increase in larger particle sediment seen at approximately 1450 AD in the three cores identified above. It is also possible that the lens seen in SB15_21 corresponds to this event, but because it is lacking a Bayesian age-depth model the linear age-depth model underestimates the age of this event. This is very probable, as accumulation rates tend to decrease with depth, so using ^{137}Cs-based accumulation rates tend to underestimate age below the cesium peak. Further ^{14}C dates around this area would resolve this question.

3.4.2.5 Sediment accretion

During the first year after sediment addition, nearly all of the plots on the augmentation site still showed feldspar on the surface, indicating negligible sediment accumulation. By one year after the sediment was added, an average of 0.5 mm of sediment had accumulated; this average was driven by a few plots with 2-3 mm of sediment accumulation, but most plots still had feldspar showing on the surface. Sediment slowly continued to accumulate until there was an average of 5.9 mm of sediment on top of the feldspar layer 62 months after sediment addition, an average accumulation of 1.2 mm/yr. There was a very wide range in accumulation, with a few plots showing none while one plot showed 23 mm. At the control site, the average sediment depth was 14.3 mm one year after sediment was added to the augmentation site. After this rapid increase the first year, accumulation decreased, with sediment accumulation reaching 18.9 mm at 62 months, an average of about 3.9 mm/year at the control site.

The mean accretion rates with standard errors by each radioisotope method of measuring accretion and across all methods from the control site and the augmentation site is shown in Table 3.2. For ^{137}Cs, the mean accretion rate is 3.9 ± 0.9 mm yr^{-1} at the control site, and 2.9 ± 0.8

mm yr^{-1} at the augmentation site. For $^{210}Pb_{ex}$, the mean accretion rate is 2.5 ± 0.6 mm yr^{-1} at the control site, and 3.3 ± 0.8 mm yr^{-1} at the augmentation site. For radiocarbon (^{14}C), the mean accretion rate is 1.8 ± 0.4 mm yr^{-1} at the control site, and 1.6 ± 0.1 mm yr^{-1} at the augmentation site. For total mean accretion rates (as determined from ^{137}Cs, $^{210}Pb_{ex}$, and ^{14}C dating), the mean accretion rate is 2.7 ± 1.1 mm yr^{-1} at the control site, and 2.6 ± 0.9 mm yr^{-1} at the augmentation site, with consistency between control and augmentation sites and radiometric methods.

Comparison of vertical sediment accretion rates by method of collection and with reference to before or after application of the augmentation sediment layer can be seen in Figure 3.7. While there are smaller dissimilarities between accretion rates at both the control and the augmentation site before the sediment layer was added, the largest contrast can be seen in feldspar mean accretion measurements that were taken after the augmentation sediment layer was added to the site. Sediment accretion in the control site after sediment was added was similar to the ^{137}Cs accretion rates, whereas accretion in the augmentation site was much lower than the ^{137}Cs accretion rates, although there was a lot of variability among samples in the post-augmentation data.

3.4.3 Permutational Multivariate Analysis of Variance

PERMANOVA tests can be seen in Table 3.1. One test compares grain size samples for the top 10 cm of each of the six cores. The second test compares grain size samples for the top 10 cm of each of the six cores to all the surface grain size samples from the augmentation site. The last test compares the bottom portion of three cores in which sand represents >20% (SB15_09, SB15_11, SB15_20) to the surface grain size samples from the augmentation site. A multivariate dispersion model was performed to test whether the groups had homogenous dispersion. For the first model (the top 10 cm of each core), the multivariate dispersion model showed that groups had homogenous dispersion, therefore suggesting that the result is indeed driven by differences in the centroids. The null hypothesis of homogenous dispersion was not rejected for models 2 and 3. However, this could be due to the unbalanced nature of our sample groups (Anderson 2001).

PERMANOVA tests comparing the top 10 cm of each core reveals a lack of significant differences between the cores ($p = 0.073$, $R^2 = 0.38$). However, PERMANOVA tests comparing the top 10 cm of the cores to the newly added augmentation sediment yielded significant differences ($p < 0.001$, $R^2 = 79\%$ of the variation in distances explained by the groups). Similarly, PERMANOVA tests comparing the core segments with sand >20% (SB) to the augmentation sediment layer yielded significant differences ($p < 0.001$, $R^2 = 54\%$ of the variation in distances explained by the groups).

The Pseudo-F value for the top 10 cm of cores compared to the augmentation layer is higher than the core segments with sand >20% compared to the augmentation layer (503.9 and 188.3, respectively). This larger pseudo-F value suggests that there are greater distances in our comparison between the top 10 cm of the cores and the augmentation layer, and lower distances in our comparison between the core segments with sand >20% and the augmentation layer sediment material. These differences are visualized in Figure 3.6, which shows the centroids of the augmentation layer compared to the top 10 cm of cores as well as the core segments with sand >20%. An important conclusion that can be drawn from the statistical analysis is that the augmentation sediment is significantly coarser in terms of sand content than even the most coarse natural sediments found in the lower portions of the cores.

3.5 Discussion

3.5.1 What is the sediment grain size and augmentation sediment thickness variability over the surface of the newly augmented salt marsh platform?

Sediment clay, silt, and sand contents from the augmentation sediment layer ranged from 0.0 – 51.6%, 0.0 – 58.0%, and 2.1 – 100.0%, respectively (Figure 3.2). Two months after the sediment was added, 80.1% was sand, 10.7% clay, and 9.2% silt. The sand fraction increased to 89.0% at 62 months, and although this is higher than at two months, the sediment remains dominated by sand. With 80% of the added sediment being sand (at two months after sediment addition), there wasn't much opportunity for the sediment to consolidate, shift, or erode into tidal creeks.

Similarly, augmentation sediment thickness at the surface changed little during the five years of monitoring. Two months after sediment was added to the augmentation site, the added sediment depth had a depth of 25.1±1.1 cm, while at 62 months, sediment depth was 23.9±1.2 cm (Figure 3.3). However, although there was overall little change in average thickness, there were changes in its spatial distribution. Early on, the thinnest sediments were on the eastern half of the augmentation site, while the southwest portion had much thicker sediments. This pattern was reinforced over time, with the eastern half of the site typically losing 1 to 5 cm, while the southwestern portion gained sediment, mostly 0 to 3 cm but some portions gained 3 to 6 cm. Although some areas experienced moderate changes in sediment depth, averaged across the entire site, there was only a modest decline of about 1 cm from 2016 to 2021. Overall, sediment thickness changed little in the five years after sediment addition. This result was surprising because sediment thickness was expected to decrease substantially after it was added due to consolidation, but the high sand content resulted in little change.

3.5.2 How does augmentation sediment grain size compare to recent and prehistoric sediment at the augmentation and control site?

Applied sediments were markedly different from prehistoric sediments at the site. Grain size analysis completed on six cores (three from the control site and three from the augmentation site) show that grain size at the top 5 cm of the cores average at 11% clay, 77% silt, and 10% sand. When comparing the top 10 cm of the six cores, we see that historical grain size values are fairly consistent across cores, between the augmentation and control sites (Table 3.1). Similarly, there is consistency between our three longer cores around 1450 AD and older (Figure 3.5). Maximum sand percentage in any sample analyzed was 76%, but these high levels of sand occur only in small lenses or below 1 m in depth where habitat may or may not have been salt marsh as it is today. However, when compared to post-augmentation grain size measurement taken from Feb – Jun 2016, we see averages of 9% clay, 10% silt, and 83% sand.

These differences demonstrate that sand concentration post-augmentation greatly exceeds sand concentrations at the top of the cores in both the control and pre-augmentation sites, as well as exceeds any sand concentration obtained in analysis of all cores covering a history of 1500 years of accretion. Additionally, we see differences that are statistically significant when comparing the augmentation layer to the top 10 cm of the cores, as well as the core segments with >20% sand concentrations. These results confirm that the sediment material in the augmentation layer is distinct from the grain size material of the natural environment found at any point in time at the site.

While the disparity between augmentation grain size and natural grain size is concerning, this record of rapid environmental change demonstrates a potential capacity for recovery. By plotting the grain size results by age, we can estimate that the lenses seen in cores SB15-09, 11, and 20 are an event previously identified as an abrupt subsidence event likely due to a tectonic event caused by the nearby Newport-Inglewood/Rose Canyon fault system (Leeper et al., 2017). Leeper et al. identify this event as having occurred from approximately 1320 AD to 1590 AD. This matches the increase in larger particle sediment seen at approximately 1450 AD in the three cores identified above. Similarly, changes between a sand-dominated grain size environment and a silt-clay dominated grain size environment have occurred in the past on estimated timescales of 10 – 30 years, as well as historically in the early phases of marsh formation (Figure 3.5). This indicates that although the applied augmentation sediment has a very different composition from the natural sediment seen in this record, the potential for recovery to a more typical state exists.

3.5.3 How different is the accretion rate of the augmentation layer compared to the natural accretion seen historically in the environment?

Historically, accretion rates at Seal Beach NWR were fairly typical of the region (Brown et al., 2022; *in-prep*). The accretion rates for ^{137}Cs and ^{210}Pb are average, if on the low end, for North American salt marshes, which can see vertical accretion anywhere from 1 mm yr^{-1-} to 10s of mm a year in high-accreting zones (Kirwan et al., 2016). Because Seal Beach is cut off from freshwater input, all accretion must be from additions of marine sediment, aeolian input, intra-marsh redistribution of mineral material, or organic input. The Mediterranean climate of Southern California means precipitation and stream flow tend to be intermittent.

Similarly, by taking an average of long-term accretion rates from ^{14}C dates, estimated average sediment accretion at the augmentation site is 1.6 ± 0.1 mm yr^{-1}, and 1.8 ± 0.4 mm yr^{-1} at the control site (Table 3.2). These values are typical for accretion rate measurements obtained from ^{14}C-dating in North American salt marshes, especially those on the Pacific Coast. These accretion rates are, however, low in comparison with accretion rates obtained from ^{137}Cs, ^{210}Pb, or modern monitoring methods such as feldspar marker horizons (Figure 3.7). This is due to the time span this analysis covers. Natural processes such as sediment compaction, local subsidence and organic decay make ^{14}C rates of accretion an underestimate of current rates, and therefore unsuitable for comparison use in modern ecosystem monitoring.

The sediment addition during the augmentation obviously exceeded natural accretion rates. In contrast, sediment accumulation on top of the added sediment in the augmentation site averaged only 1.2 mm yr^{-1}, which is lower than the long-term average accretion rate of 1.6 mm yr^{-1} and much lower than the accretion rate of 2.9 mm yr^{-1} over the past 60 years. By contrast, the average accumulation in the control site was 3.9 mm/year. This suggests that sediment accumulation in the control site was three times higher than the augmentation site, and more similar to salt marsh accumulation rates found in the southern California region (Thorne et al. 2018).

The difference in accretion rates between the augmentation site and the control site may be explained by differences in vegetation cover. The control site consistently had dense vegetation while the augmentation site remained largely unvegetated during the monitoring period. Post-sediment addition, there was very little sediment accumulation for the first two years at the augmentation site, while the control site had consistently higher rates of sediment accumulation, with the highest rates deposited during the first year after sediment addition.

Given the time-proximity to the sediment addition, it is possible that the control site received these sediments over the first year from the sediment addition at the augmentation site. Thorne et al. (2019) reported a deposition of 4-5 mm of sediment on the feldspar at the control site immediately after sediment addition in the augmentation site, which does suggest an influence of the sediment addition on the control site.

Our results suggest that modern sediment accretion rates at the augmentation site are lower than the natural accretion seen historically in the environment, with mean accretion values of 1.2 mm yr^{-1} and 2.6 mm yr^{-1}, respectively. Conversely, sediment accretion rates at the control site are higher than the long-term historical values, with 3.9 mm yr^{-1} and 2.7 mm yr^{-1}, respectively, although the same as the accretion rate over the past 60 years (3.9 mm yr^{-1}). However, these values this may be influenced by the sediment addition project, particularly at the Control site.

3.6 Conclusion

Climate change and SLR pose one of the greatest risks to coastal wetlands in the near term. Historical observations show that tidal wetlands can tolerate some levels of SLR; however, anticipated rates of SLR, disruptions of natural sediment sources and coastal squeeze are important challenges to understand and manage future salt marsh trajectories. Sediment augmentation projects provide a potential opportunity to mitigate the impacts caused by SLR. Additionally, this study provided a unique opportunity to understand impacts of sediment addition within the long-term dynamics of the marsh and shorter-term response to the augmentation at a Pacific Coast marsh. The artificial application of thin-layer sediment at Seal Beach NWR marsh is one of the first attempts to maintain marsh habitat with sediment enrichment along the Pacific coast of the USA. Through the combination of comprehensive monitoring and sediment cores, we hope this has enhanced understanding of marsh responses to treatments, informing more effective future augmentation projects.

The impacts of sediment addition in the short-term (1 year after application) were generally negative on the ecological community of the ecosystem, with the loss of vegetated areas. The augmentation site, prior to sediment addition, had a diverse assemblage of plants with generally high cover, but low stature; therefore, it was not suitable habitat for endangered nesting birds (Figure 3.1b). However, the vegetation community 1-year post sediment addition had not recovered and there were large barren high-elevation areas along the marsh platform. Five years post-augmentation, the landscape still lacks vegetative diversity and abundance, especially when compared to the control site, indicating a slower than anticipated recovery of the site (Figure 3.1c). This slow vegetative regrowth undoubtedly impedes the short and long-term recovery of invertebrates and other animal species within the marsh, due to factors such as lack of vegetation cover and low soil organic content.

These delayed recovery for vegetation may be due to changes in soil composition from the original soil to the newly added dredged material. Due to challenges in obtaining augmentation material, the grain size of the new dredge material was coarser, with \much less silt and clay than the pre-augmentation material or the control site. Once the sediment was applied on the augmentation site, the overall sediment grain size was low in silt and clay content (16%). Additionally, the lack of access to sediment sources such as fluvial networks, storm events, or more likely organic matter accretion coupled with the newly developed supratidal elevation regions also reduced the tidal inundation period across the augmentation site. Although tides provide the necessary source materials for salt marsh platforms to develop, the frequency, duration, and depth of tidal inundation define salt marsh characteristics. An example of this is how increased inundation allows for higher rates of sediment deposition, while increased frequency and duration limit plant productivity (Wiberg et al., 2019; Morris et al., 2002; Kirwan and Megonigal 2013). Furthermore, reduced tidal inundation can influence salinity levels. High salinity concentrations impede plant communities from establishing, reducing the ability of crustaceans, mollusks, and other biota to move in and thrive, preventing the microtopographic salt marsh landscape from developing and aiding in marsh resilience (Whitcraft and Levin 2007; Sievers et al., 2019).

This project highlights the important role that sediment characteristics can play in salt marsh recovery rates. Specifically, grain size sediment materials need to match the natural environment. Our results show that, due to the differences in sediment grain size of the parent material compared to the newly added outsourced material, the sand fraction is higher and the organic content of the sediment much lower at the augmentation site. In a depositional environment like the salt marsh at Seal Beach, small particles such as silt and clay tend to make up the dominant portion of mineral material. Ideally, grain size added during thin layer sediment application to increase elevation should be similar to grain sizes seen in the past to mimic natural salt marsh conditions and promote plant growth. In addition, increasing the marsh plain (in this case by 25 cm) will result in changes to hydrology and sediment dynamics. However, in the long-term, this higher marsh plain may reduce the vulnerability to drowning by SLR.

3.7 Figures

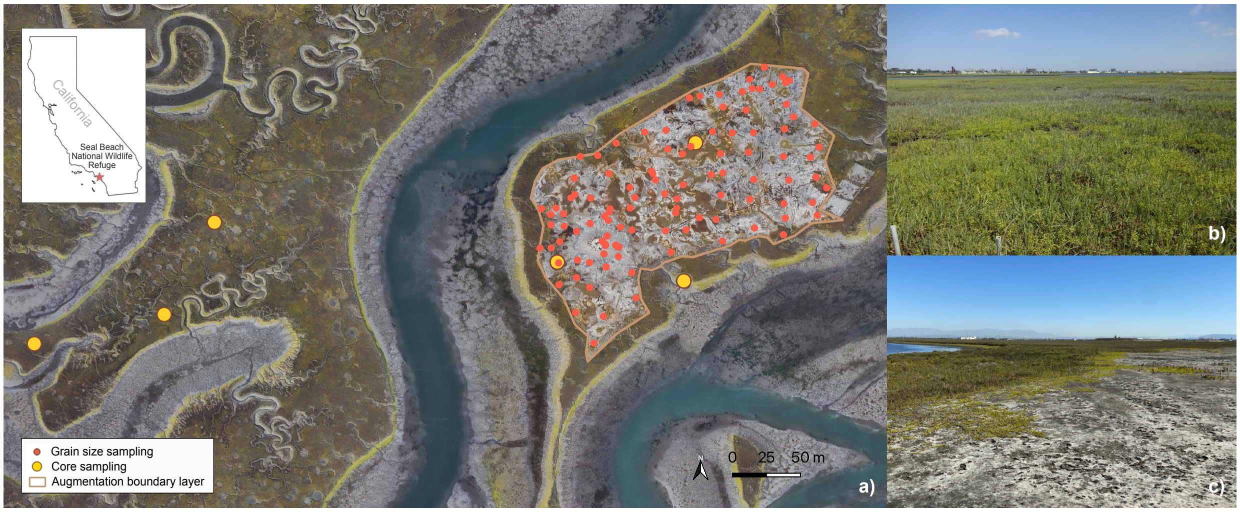


Figure 3.1 Site map of Seal Beach National Wildlife Refuge

Site map of Seal Beach NWR with a) surface grain size sample locations and core extraction locations in the augmentation and control site, b) a picture of the marsh plain at the augmentation site in July 2015, and c) a picture of the marsh plain at the augmentation site in March 2022. Background: google satellite in QGIS.

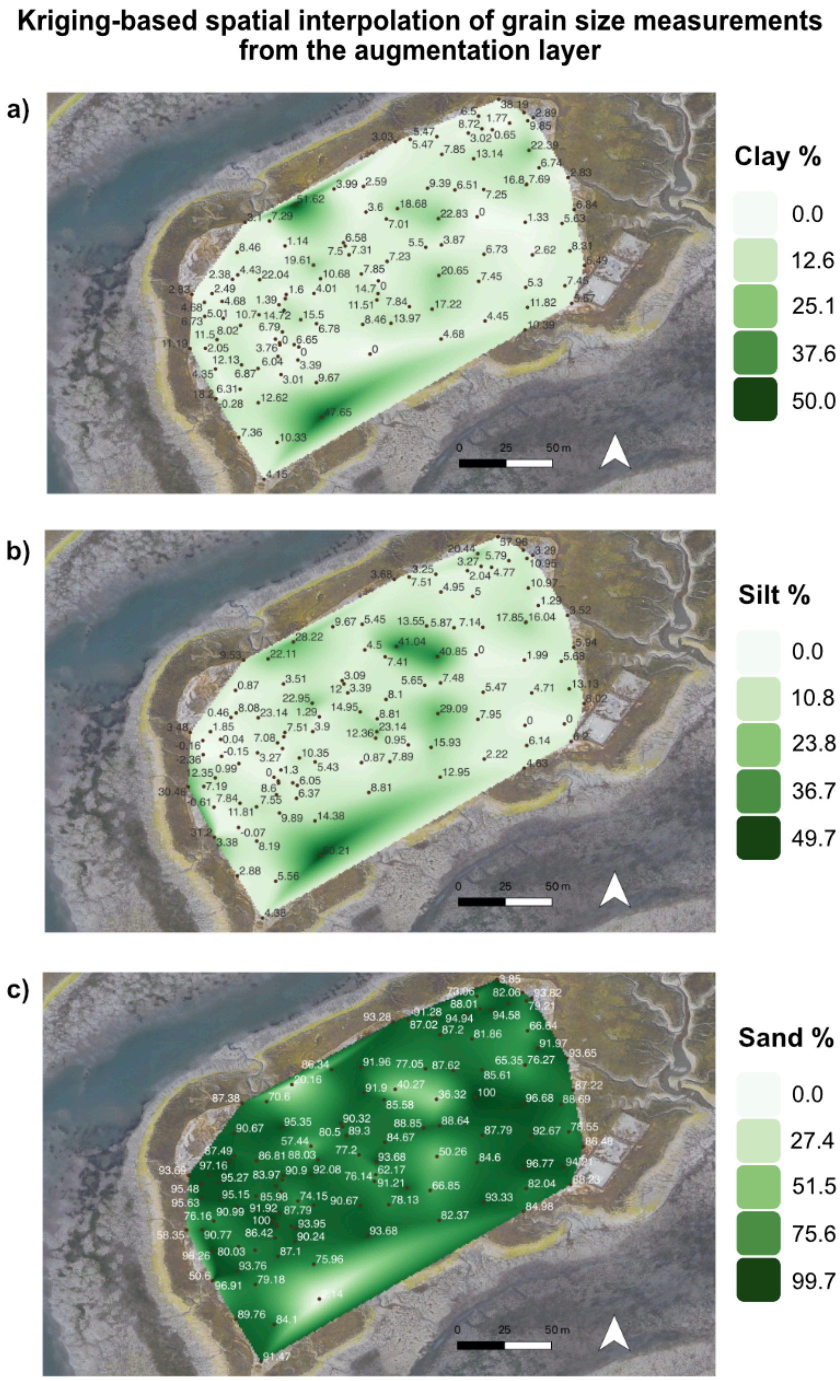
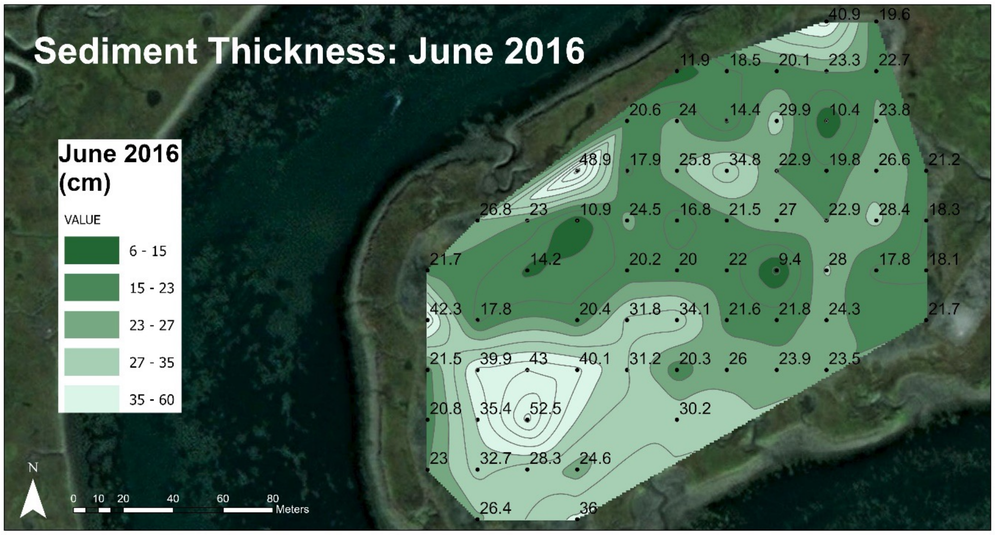
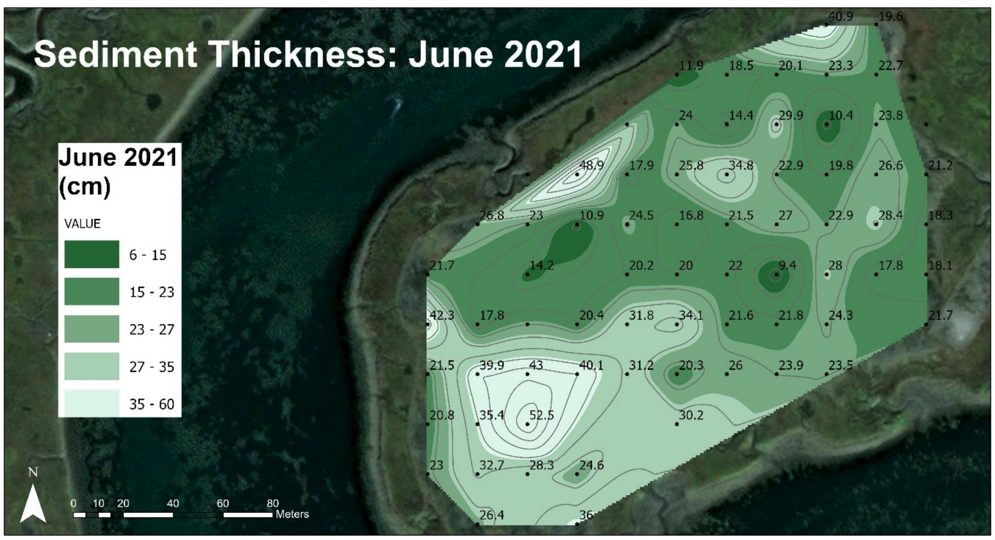


Figure 3.2 Kriging-based spatial interpolation of augmentation layer

Spatial distribution patterns of grainsize (a) clay (b) silt and (c) sand in augmentation layer, generated by kriging interpolation methods at Seal Beach NWR, California. Background: google satellite in QGIS.



a)



b)

Figure 3.3 Sediment thickness map

Map of sediment thickness on Augmentation Site for a) 2 months after sediment was added, June 2016 and b) 62 months after sediment as added, June 2021. Data from the sediment stake grid.

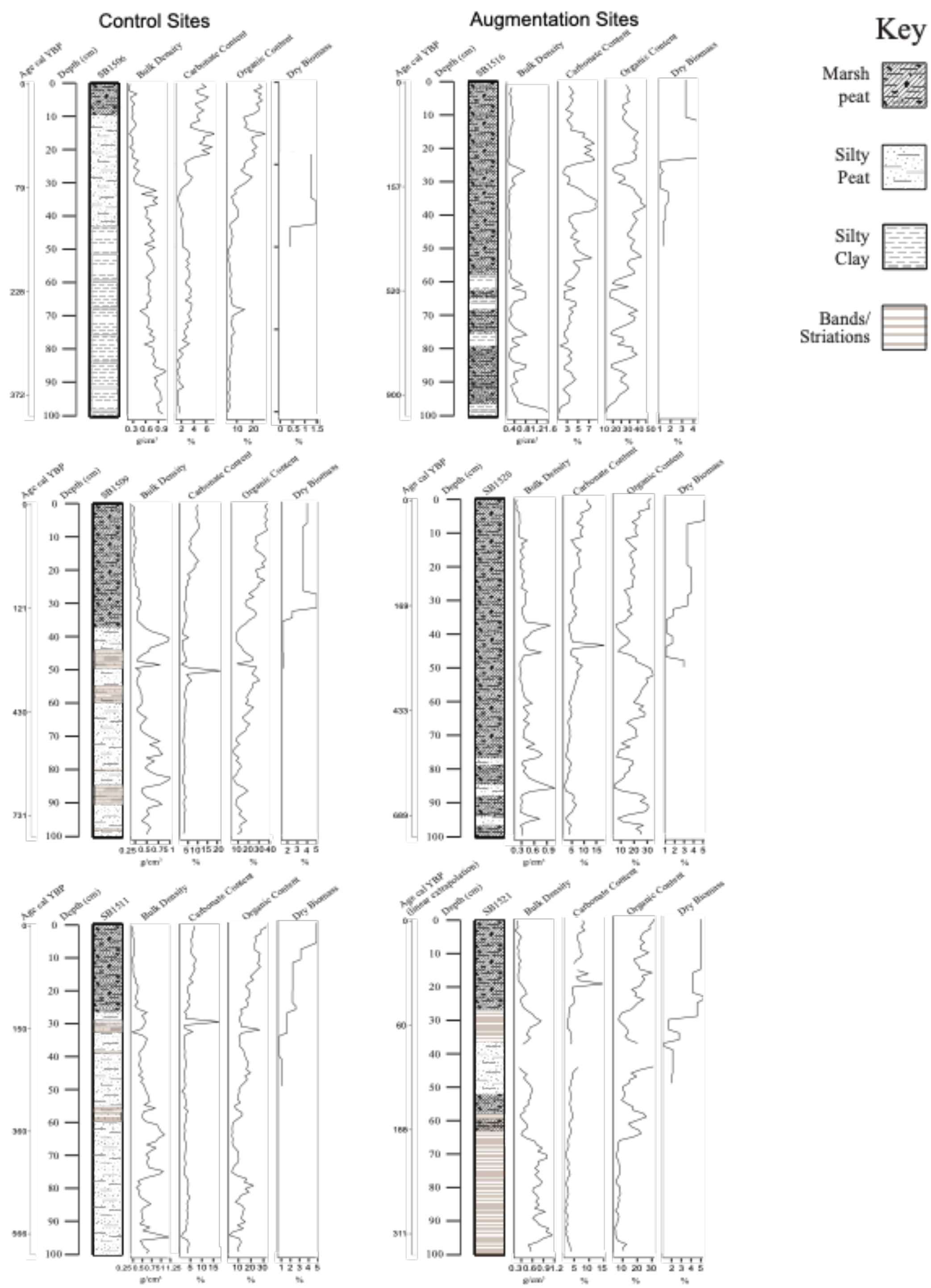


Figure 3.4 Core stratigraphy

Core stratigraphy, LOI variables (bulk density, carbonate percent, organic percent), and biomass concentrations placed against Depth (cm) and Age.

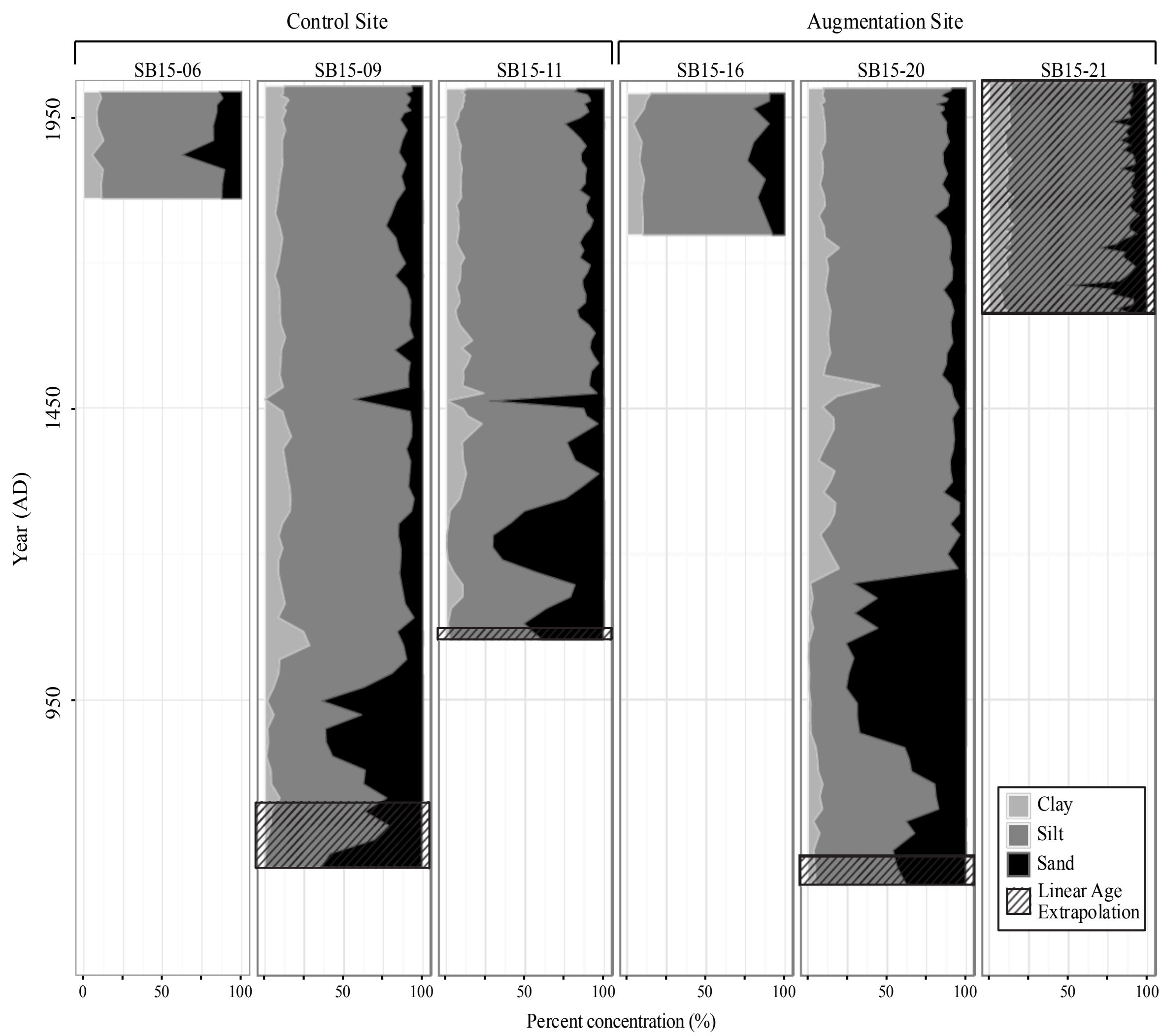


Figure 3.5 Core grain size

Grain size analysis by time for the control and augmentation sediment cores.

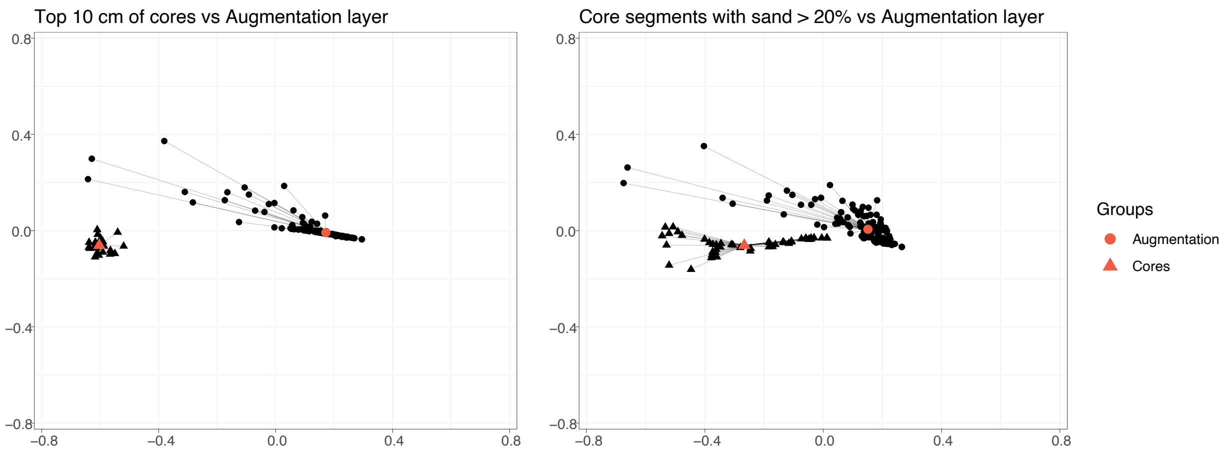


Figure 3.6 PERMANOVA distance matrix

PERMANOVA centroid and distance matrix results visualized for the top 10 cm of cores compared to the augmentation sediment layer, and core segments with sand > 20% compared to the augmentation sediment layer.

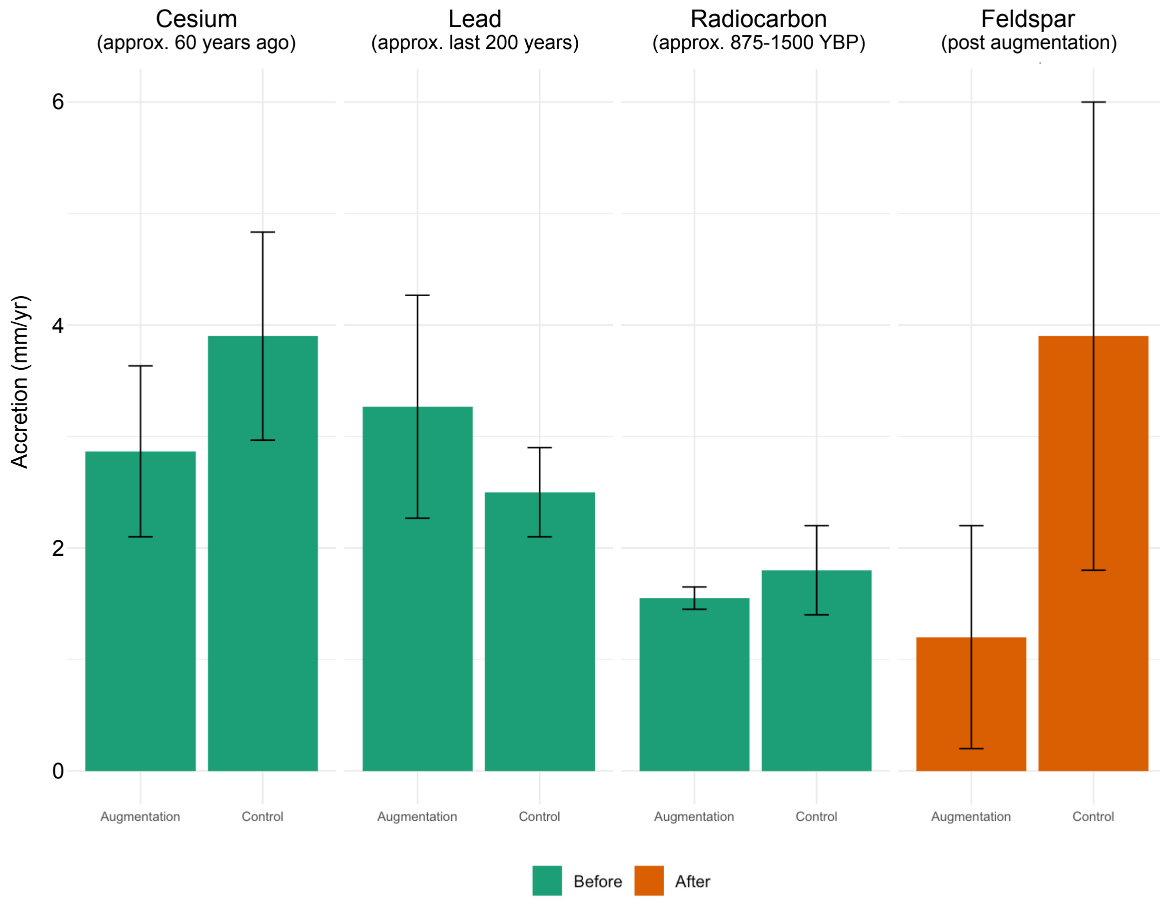


Figure 3.7 Accretion rates by method

Comparison of vertical sediment accretion rates by method of collection. Before signifies sediment accumulation before application of the augmentation layer; after refers to sediment accumulation after the augmentation layer was applied.

3.8 Tables

ID	df	SS	MS	Pseudo-F	P(perm)	Perms
Top 10 cm of cores (6)	5	176.85	35.371	2.1841	0.073	999
Res	18	291.5	16.194			
Total	23	468.35				
Top 10 cm of cores (6) to augmentation	1	196482	196482	503.88	0.001	999
Res	136	53031	390			
Total	137	249514				
Core segments with sand > 20% (3) to augmentation	1	94662	94662	188.26	0.001	999
Res	160	80450	503			
Total	161	175112				

Table 3.1 PERMANOVA results

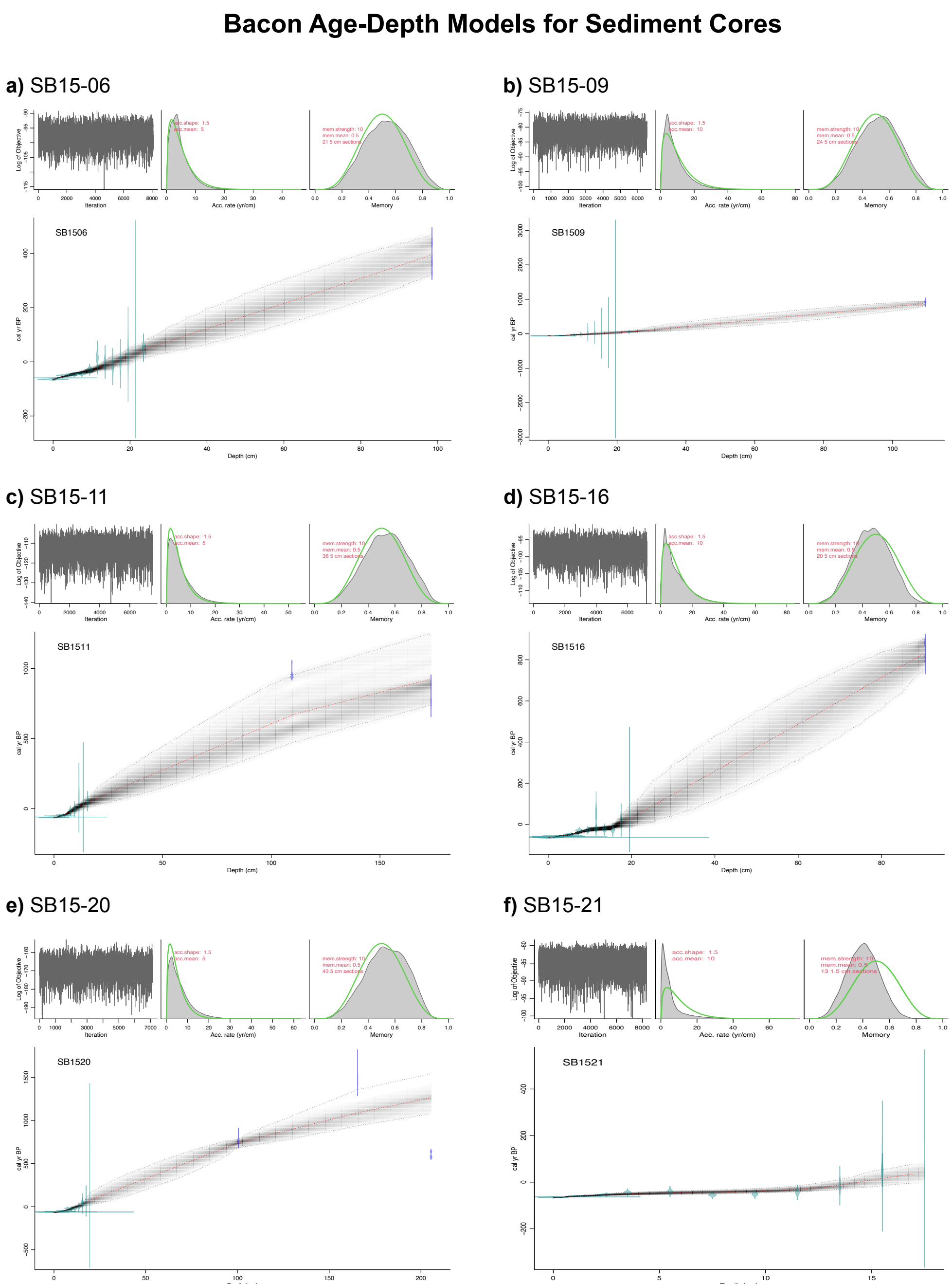
PERMANOVA results table for grain size comparisons include Top 10 cm of cores (6 cores used), Top 10 cm of cores (6 cores used) to the augmentation layer (113 surface samples), and Core segments with sand > 20% (3 cores used) to the augmentation layer (113 surface samples). df: degrees of freedom; SS: sum of squares; MS: mean sum of squares; Pseudo-F: F value by permutation, P(perm): p-values based on more than 999 permutations (the lowest possible p-value is 0.0001); Perms: number of permutations.

Site	Mean Accretion ($mm\ yr^{-1}$) ^{137}Cs (n)	^{210}Pb (n)	^{14}C (n)	All
Control Site	3.9 ± 0.9 (3)	2.5 ± 0.6 (3)	1.8 ± 0.4 (3)	2.7 ± 1.1 (9)
Augmentation Site	2.9 ± 0.8 (3)	3.3 ± 0.8 (3)	1.6 ± 0.1 (2)	2.6 ± 0.9 (8)

Table 3.2 Mean accretion results

Control site and Augmentation site mean accretion rates in $mm\ yr^{-1}$ with standard errors by each method of measuring accretion and across all methods based on ^{210}Pb and ^{137}Cs, and radiocarbon dates based on ^{14}C, for all sampling sites. Accretion rates obtained from USC. Radiocarbon dates obtained from UC Irvine Keck Radiocarbon lab.

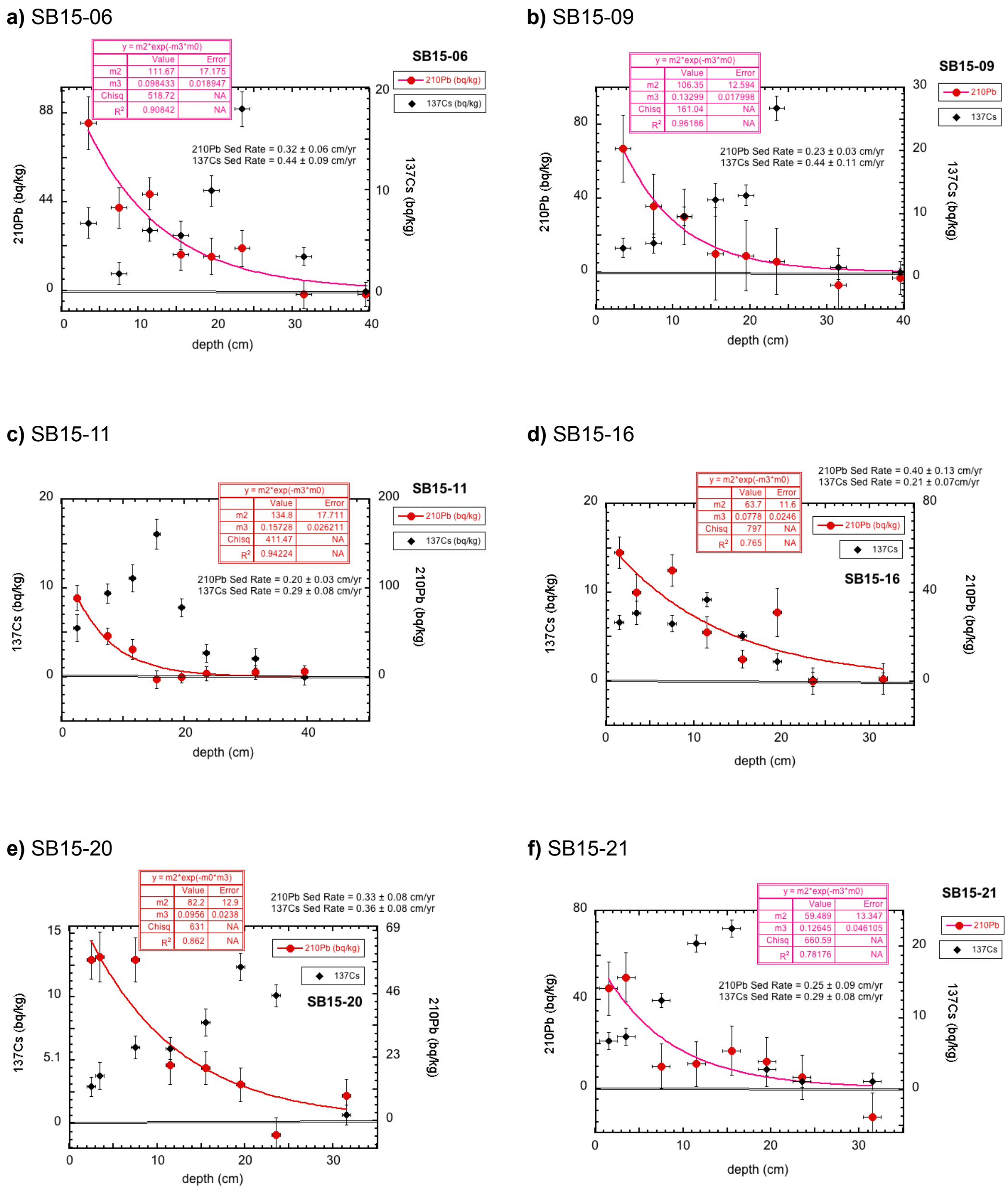
3.9 Appendix



Appendix Figure 3.1 Sediment core age-depth models

Bacon Age-Depth Models for sediment cores a) SB15-06, b) SB15-09, c) SB15-11, d) SB15-16, e) SB15-20, and f) SB15-21.

Lead and Cesium Curves for Sediment Cores



Appendix Figure 3.2 Lead and Cesium curves

Lead and cesium curves for sediment cores a) SB15-06, b) SB15-09, c) SB15-11, d) SB15-16, e) SB15-20, and f) SB15-21.

Core	Depth (cm)	14C Age (BP) Uncalibrated	Error (BP)	Age (YBP)	Error (YBP)
SB15-06	99	340	15	380	78
SB15-09	110	995	20	931	27
SB15-11	110	1060	15	956	25
SB15-11*	174	870	40	781	106
SB15-16	91	925	15	875.5	36
SB15-20	101	865	20	776	83
SB15-20	166	1620	60	1502	126
SB15-20*	206	640	15	589	50

Appendix Table 3.1 Radiocarbon table

Radiocarbon results, reported as uncalibrated ^{14}C age before present (BP) from the University of California, Irvine Keck-Carbon Cycle AMS facility, as well as calibrated years before present (YBP).

3.10 References

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4. The politics and economies of coproduction: Assessing the climate science-policy interface in California

4.1 Abstract

The challenges of climate change have added pressing urgency to the development of effective science-policy interfaces. Scholars of the science-policy interface have long argued that greater direct collaboration between scientists and decision-makers is necessary to ensure successful policy responses. Several different models of science-policy interaction have been proposed in order to assist in managing the impacts of climate change: coproduction, boundary organizations, and climate services. This paper catalogues and assesses efforts to implement these models to manage natural resources in the context of climate change in California, USA. Drawing from original participant-observer and interview research, this paper examines the practices and some consequences of production and circulation of climate change science in the context of California. We find, first, many examples of scientists and decision-makers working together to produce useful information, but, second, too few cases where this information has led to transformative adaptation actions. Third, we highlight how science-policy models emerge within, and can be constrained by specific decision-making culture and political economic context. We demonstrate how these models can reinforce the localization and devolution, and the outsourcing and privatization of knowledge. We show that these science-policy models reflect and reproduce regional contexts that can limit opportunity for replicability. Thus, we argue the importance of viewing different coproduction efforts as a collective, rather than closed systems. We also highlight the strength of boundary organizations at local and national scales that have the ability to minimize regional spatial and temporal variability. Lastly, we explore the use of

educational awareness of science practitioners at the early stages of their careers to implement these models effectively. We discuss successes and areas of improvement for boundary organizations dedicated to engaging in and promoting climate adaptation initiatives and research.

4.2 Introduction

There is a need to better integrate climate science and climate adaptation science into management and planning of environmental resources. Research into understanding what makes knowledge relevant in management and policy contexts is not a new field (Cash et al., 2006; Lemos and Morehouse 2005; Vaughan and Dessai 2014). While scholars and practitioners have increasingly touted the benefits of co-creating meaningful knowledge, or actionable science, in decision making contexts, the 'how' of co-producing knowledge in practice is less clear (Mach et al., 2020; Arnott et al., 2020; Goodrich et al., 2020; Dewulf et al., 2020).

While there are diverse and creative ways in which researchers and practitioners have engaged in knowledge creation, three models have been the focus of most science-policy efforts: coproduction, boundary organizations, and climate services. Norström et al. (2020) describe these efforts as "iterative and collaborative processes involving diverse types of expertise, knowledge and actors to produce context-specific knowledge and pathways towards a sustainable future". These efforts anticipate that more actionable forms of scientific information about climate impacts will contribute to better climate change adaptation decisions.

Despite increased efforts in recent years, it's been argued that the political economy places restrictions on our ability to use the science in this capacity (Turnhout et al., 2020; Owen and Pansera., 2019). Additionally, there is a large gap and need in the literature for evaluation of successful coproduction efforts. This can be a difficult task, given the complexity and context-specific nature of actionable science, and the process of tracing and defining how and when knowledge moves from actionability to tangible impacts (Mach et al. 2020). Bremer and Meisch

(2017) urge self-reflexive transparency when using coproduction concepts to provide clarity and expand the concept of coproduction to be utilized in new ways.

This paper aims to provide an analysis on knowledge coproduction efforts in management settings to understand what makes knowledge relevant in management and policy contexts. To do this, we use participant observations, interviews, and analysis of documents. At the onset of our study, we looked at how the three models have been applied in the California context to examine the success, or lack thereof, regarding the integration of climate change science and climate adaptation science into planning and management, with a focus on coastal zone environments. More recently, we used the Southwest Climate Adaptation Science Center (SW CASC) as a case study to examine their approaches and results of integration efforts in science to policy through initiatives and educational training efforts. In addition, we explore the use of educational awareness of science practitioners at the early stages of their careers to implement these models effectively.

Throughout the paper we look at examples of different applications of these models in the context of California to see if geographical regionalism and political context provide a unique situation when implementing these models. Additionally, we explore an avenue forward based upon the integration of coproduction, boundary organizations, and climate service providers at the earliest inception of the effort. This framework stresses the importance of delineation of climate service needs through direct communication and interconnectivity between stakeholders and climate science providers as the motivating and organizing basis of the work. Finally, we explore the impact of including early educational components in addressing an application of these models.

4.3 Theorizing the science-policy interface

Over the last two decades, scholars of science and society have tried to understand what defines success in science-policy efforts. The science-policy literature argues that useful science should play a transformative role in creating and implementing policies to address environmental issues (Lemos 2015). This is in contrast to linear models of science which assume that decision-makers can use scientific information that is produced independently from those decisions it is used to influence (Cash et al., 2006). As researchers and practitioners work towards making knowledge more usable, as well as creating more tangible project outputs, the struggle with the complexities involved in collaborative and transdisciplinary research remains (Kirchhoff et al., 2013). Djenontin and Meadow (2018) highlight a growing body of research suggesting that the driving force behind success in coproduction is the direct connection and collaboration between researchers and practitioners (Mach et al., 2020; Kirchhoff et al., 2013; Lemos and Morehouse 2005; Lemos et al., 2012; Reed et al., 2014; Roux et al., 2017). Nordström et al. (2020) highlights another perspective, suggesting that there is no single approach for success, and that the collaborative nature of coproduction, which involves engagement across domains and disciplines, can be as important for the pursuit of sustainability as the production of knowledge. While coproduction promises to better address contemporary sustainability challenges, challenges arise when definitions are diverse or contradictory.

There are three main models that have been utilized to integrate climate science and climate adaptation science into management and planning of environment resources. The first, *Coproduction*, involves substantive interactions between producers and users of knowledge that results in knowledge that fits decisions contexts (Mach et al., 2020). Researchers argue that coproduction collaborations extend from the framing of research questions and trust building, to defining the methodology and analyzing results (Cash et al., 2006; Dilling and Lemos, 2011; Bremer and Meisch, 2017; Norström et al., 2020). Additionally, coproduction efforts extend beyond producing knowledge production, by also developing capacity, building networks, and fostering social capital (Norström et al., 2020). Coproduction will not only rework relationships between scientists and decision-makers, thereby facilitating effective climate change decisions, but also transform the scientific product itself.

The next model includes *boundary organizations*, which are institutional arrangements – virtual, financial, or physical – that link scientists and decision makers, often with the assistance of intermediaries or knowledge brokers that work between science and policy (Hoppe et al., 2013; Mauser et al., 2013; Lemos et al., 2012). Cash et al. (2003) suggest that such organizations facilitate extended relationships and collaborative outputs through four necessary functions: convening constituents, translating problems and outputs, collaboration, and mediation between members. These organizations are understood to be good at connecting useful science with its users "like links in a chain" (Lemos et al., 2014). This chain approach seeks to resolve mismatches between the number and scale of potential users and producers, allowing flexibility among and between the different nodes of the chain. Additionally, the interest in boundary institutions has given rise to the profession of 'boundary spanners'. Goodrich et al. (2020) suggest this effort is led by individuals who work independently or within a boundary organization, engaging in cross-disciplinary efforts that involve mediating, bridging, and brokering knowledge. This engagement can be between individuals or organizations that generate knowledge and end-users who apply the knowledge in decision-making contexts. Boundary spanning professionals are also beginning to share lessons learned from their own experiences in their respective fields to better understand best practices and outcomes.

Recently, *climate services* have been proposed as one way of conceptualizing the desired output of coproduction and boundary organizations (Vaughan and Dessai 2014; Nost 2019; Owen et al., 2019). Climate services are defined as the “timely production, translation, and delivery of useful climate data, information, and knowledge for societal decision-making” (Vaughan et al., 2016). Climate services, such as observational data sets, online decision tools and descriptive visualizations of forecasts and predictions, are packages of information intended to be timely and tailored to the decision-contexts in order to reduce the risks – or enable societies to take opportunities – that arise due to climate change (World Meteorological Organization, 2011). These services are created with the intention of providing the tools necessary to improve society's resilience to climate-related hazards, allowing users to better manage the risks and opportunities arising from climate variability and climate change (Hewitt et al., 2012). Brasseur and Gallardo (2016) argue that the success of climate services has been hampered due to inappropriate format in which the information is provided, and the inadequate business model adopted by climate services. The authors suggest that “centers should host within the same center a diversity of staff including experts in climate science, specialists in impact, adaptation, and vulnerability, representatives of the corporate world, agents of the public service as well as social managers and communication specialists” to create a successful climate service model.

With the increased utilization of these models comes greater research about the significant impacts of coproduction, boundary organizations and climate services, as well as their affectability. Brasseur and Gallardo (2016) demonstrate that while there are individual instances of successful collaboration, climate information still very rarely informs adaptation decision-making. This is perhaps because these efforts are still driven by the ‘producers’ or ‘suppliers’ rather than consumers (Lourenço et al., 2016). Additionally, many projects and interventions using the interdisciplinary framework of coproduction, transdisciplinarity, science-policy interface, democratization of expertise, or knowledge brokering have been found to fail to live up to their stated objectives (Turnhout et al. 2020). Still, coproduction is advocated as a popular model for creating usable knowledge when compared to the traditional unidirectional flow of information from researchers to policy makers (Mach et al. 2020).

Research on the contexts of considering the politics of the social contexts within which science is co-produced remains under-explored in the existing science-policy literature. And the research that does exist oftentimes looks at these models independently, as closed systems, producing knowledge that can be isolated from the spatial, temporal, and social contexts from which it emerges. To address this, in this study we examine the success and potential pitfalls of integrating these models collaboratively through examples of coproduction efforts, and a case study of the initiatives created by the SW CASC.

4.4 Climate science-policy in California and the Southwest CASC

California has a diverse assemblage of local, state, and federal bureaucrats, academics, private consultants, and non-governmental organizations (NGOs) working on environmental science, conservation, and climate change adaptation. It has passed some of the strongest climate polices in the US (Franco et al., 2008), and is regarded as a place where progressive environmental action can gather regional and statewide support while having the ability to also influence national and global initiatives. As such, it provides a good case study for looking at a diverse range of actors and instructions applying various science-policy models in unique ways. Similarly, we can see the communicative process as there are many collaborative entities within the state.

On the science-producer side, the US National Oceanic and Atmospheric Administration (NOAA) initiated the Climate Assessment for the Southwest (CLIMAS) program. CLIMAS seeks to improve adaptive capacity to climate change and variability through the provision of regional climate information for those making adaptation decisions.[1] To enable this vision, CLIMASs focus on research engagement and integration across the natural and social sciences, and on building connections between academic expertise and diverse intermediaries. Parties to this vision include decision-makers, private consultants and non-profit organizations from a range of sectors, sharing a focus on "a certain place, state, or region."[2]

In parallel to the CLIMAS program, the US Department of Interior (DoI) United States Geological Survey (USGS) has created a network of nine Climate Adaptation Science Centers (CASCs) covering the continental U.S., Alaska, Hawai'i, the U.S. Affiliated Pacific Islands, and the U.S. Caribbean. The tenth National CASC (NCASC) headquarters is based in Reston, Virginia, and serves as the managing entity for the broader CASC networks. The CASCs operate as a collaborative, boundary organization that convenes scientists from the USGS and a network of regional academic institutions to provide climate information and methodologies for regional natural resource management – including land, water, wildlife and cultural assets.

Additionally, CASCs have made ongoing efforts that provide climate services in the form of multi-disciplinary workshops. The most recent example in California includes a series of coastal management workshops, sponsored by the Southwest CASC and held by the University of California, Los Angeles (UCLA) between 2019-2022. The introduction of the workshop engaged with coastal resource managers, non-profit organizations, federal and state agencies, and

[1] Interview 25
[2] Interview 25

university scientists to identify management issues and priorities related to the impacts of climate change on coastal ecosystems. The workshops were informed by previous research. Previous workshops led to a subsequent series of workshops focused on creating collaborative network opportunities by hosting coastal scientists, managers, and planners in an ongoing effort to facilitate science coproduction. Like CLIMAS, the CASCs include research projects driven by integrative and multi-disciplinary goals.

Working in parallel with the Federal initiatives are components of State agencies such as the California Energy Commission (CEC), a unit of the California Natural Resources Agency. In addition to other tasks, until recently the CEC channeled funding for the California Climate Assessment, under a mandate to provide information about climate change mitigation and adaptation for policy makers and managers.[3] Through this process they facilitated research that drew together a wide variety (including private consultants) of scientists together with resource agencies. Californian institutions using climate information for their decisions include departments of the Natural Resource Agency, such as the Department of Water Resources and the Department of Fish and Wildlife. Additionally, the California Environmental Protection Agency (CalEPA) has led in creating and implementing some of the most progressive environmental policies in the US, including the Global Warming Solutions Act (AB 32) and launching the Green Chemistry Initiative. These departments have funded and co-produced major studies of climate impacts.

A variety of non-state and non-university actors are increasingly involved as intermediary consumer-producers-cum-knowledge brokers in this science-policy assemblage. In California,

[3] Interview 1

organizations such as the Nature Conservancy and Point Blue are producers, consumers and appliers of scientific information, as managers of significant amounts of land.[4]

4.5 Methods

Conducting research in California, Arizona, and Washington DC from 2016 through 2022, three researchers involved in this study employed three methodological techniques and sources of information to investigate the science-policy interface: participant observation, documentary analysis and in-depth key-informant interviews. The focus of the research was the Department of the Interior's Southwest Climate Adaptation Center (SW CASC) and its efforts. One researcher (author of this dissertation) is involved in current (2019-2022) coproduction and boundary work related to the adaptation of California's coastal ecosystems to climate change, participating from this vantage point in scientific production, management, and communication activities. Specifically, she was involved in coproduction and boundary work related to the adaptation of California's coastal ecosystems to climate change and helped organize, participated in, observed, and catalogued workshop discussions. This researcher received training in the process of developing policy relevant science from the Natural Resource Workforce Development Fellowship program of the SW CASC (2019-2020). This researcher is also responsible for the final collation of study observation and notes and drafting of this chapter. Another researcher conducted interviews of scientists and stakeholders in California and elsewhere and participated in and observed workshops and conferences seeking to bring together decision-makers and scientists to refine research needs and encourage collaboration (2016-2018). She also conducted a review, contextualization and synthesis of the early study information. A

[4] Interview 17

third researcher is a Co-PI of the SW CASC and provided initial conception and ongoing supervision of the project. He has been a participant observer in SW CASC science-policy efforts and activities throughout.

Documentary analysis of existing publications and reports, written by and for the identified science providers and users, supplemented the primary field research. These sources added to the contextual account of the relations between science and policy, and were necessary for informing key interview questions, and triangulating informant responses. Participant observation and the collection and analysis of documentary evidence can begin to overcome some of the pitfalls of key-informant interviews, including 'agent inflation' (Tickell et al., 2007) where informants overstate their influence and importance.

In-depth key-informant interviews with 31 senior actors from across the spectrum of science-policy work was conducted by the second researcher; the interviews at each institution are outlined in Appendix 4.2. Rather than seeking a sample of similar informants, these interviews included influential technical experts and decision makers occupying different vantage points. Interviewees included individuals acting at regional, state, and national scales: personnel who design, implement, manage, and finance science and adaptation efforts. Interviewees were identified using snowball sampling, networking at conferences and workshops, and building on existing relationships. Key themes of interviews included the successes and limitations of their existing projects, interviewees' knowledge practices and assumptions, and the changing relationships between science and management. Interviewing continued until no significant new information was forthcoming. The data collected were summarized, transcribed and coded according to topics that unpack the changing nature of scientific and social relations, practices, and formations. These topics included: the diverse actors involved in the science-policy ecosystem; the kinds of scientific and policy work conducted; and the causes of failures and successes.

4.6 Results: How have these models been applied and to what success?

Throughout the research, participants in the science-management assemblage identified dozens of different examples where resource managers (of water, coasts, land, and plant and animal species) and scientists (of ecology, conservation biology, and climate) collaborated to inform decision-making. As one state agency employee notes, "the issue of climate change has really thrust a lot more people into that position of saying, well, I'm a scientist, but I want to help."[5] This is not the only motivating force; nonetheless, the challenge of managing resources amidst climate change and variability is central to the growth of new science-policy institutions. This section catalogues the breadth of these science-policy interactions, identifying: first, different modes of collaboration; second, instances where collaborations were successful; and third, some of the emergent challenges. We demonstrate that, although new science-policy interactions proliferate, current successful collaborations that inform decision-making are limited in their replicability.

One key way that scientists and resource managers collaborate is by developing broad or general documents, such as assessments, reports, and strategic plans. For instance, the multi-agency California Climate Assessment released its first installment in 2006 and released its current Fourth Assessment (https://climateassessment.ca.gov/) in 2018. A fifth Assessment is currently in progress. Each of the assessments had a different focus and funding mechanisms for researchers to conduct research about climate change that broadly informs policy and

[5] Interview 16

management in California. The CEC could use the Assessments to bring together scientists and managers "because we had money, and we were in charge of directing the research."[6] However, it is more difficult to know exactly how this information contributes to management decisions: because "we don't… track how things are being used."[7] In addition to state Climate Assessments, adaptation strategies and action plans also convene management agencies and scientists. These include: the California Climate Adaptation Strategy[8] that brings "together sector leads [water, fire, biodiversity, etc.]" to plan how to adapt resources,[9] and the Wildlife Action Plan that "created teams of [agency and other] scientists" to identify climate-related and other threats and stressors to habitats throughout California.[10] There also exists a portfolio of climate service products that contribute to plans, reports and assessments, including CalAdapt and the California Climate Commons.[11] For scientists working with managers on climate questions, there are "lots of pockets of money" to fund their work.[12] As one interviewee summarized: in the world of conservation management, there is "more and more" coproduction, boundary organizations and climate services.[13]

Another way to induce science-management collaboration is to contract research on a specific question. For instance, the California Department of Fish and Wildlife commissioned vulnerability assessments for vegetation and mammals in order to "get a really comprehensive view of vulnerable areas of the state."[14] This information can help identify different refugia and

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[6] Interview 1
[7] Interview 1
[8] http://www.climatechange.ca.gov/adaptation/strategy/index.html
[9] Interview 2
[10] Interview 15
[11] http://cal-adapt.org/
http://climate.calcommons.org/
[12] Interview 19
[13] Interview 16
[14] Interview 16

"inform different conservation priorities [and] land acquisition practices."[15] And yet, even in these cases where specific management problems have led to a research contract that serves a clear focus, it is still the case that resource managers and policy makers "are very, very good about planning, but they take a very hands-off approach to management."[16] Even in such narrow cases it was difficult for interviewees "to think of an example [of using science in management decisions] because it doesn't happen a lot."[17]

One successful case identified by many interviewees is a coastal salt marsh sediment augmentation project in a Southern California, implemented at Seal Beach National Wildlife Refuge (NWR). In this project, USGS and university scientists, other partners and FWS coastal land managers (the lead agency) have sought to test marsh conservation decisions (Thorne et al., 2019; McAtee et al., 2020; Sloane et al., 2021). The refuge revolves around "trying to ensure the long-term sustainability of the salt marsh" in order to protect two endangered migratory bird species.[18] But the "marsh was already being affected by subsidence and it was only going to get worse as sea levels began to rise."[19] The science-management collective embarked on a process of 'adaptive management' through sediment augmentation, with approval to experiment with actions to try to sustain the marsh. Armed with funding from state and federal sources, specific and useable science about elevation, subsidence, sediment balance and sea level rise produced by USGS and university scientists for the refuge,[20] and a willing team, they are implementing an experimental project "to try and increase the height of the marsh but allow vegetation to punch

[15] Interview 16
[16] Interview 5
[17] Interview 15
[18] Interview 13
[19] Interview 12
[20] Interview 12

through that sediment layer."[21] Three years of planning and the sediment augmentation itself, were followed by an extended research period with "five years of monitoring, … [to] gauge its success [and] all the lessons learned."[22]

There are three reasons why this example was successful and one reason why it was a failure. First, there was a clear vision with specific goals, tightly targeted funding, and a seasoned team familiar with multi-institutional work. The project emerged from a longer collaboration between managers and scientists that produced site-specific scientific information that informed funding applications for management actions and further research. Such site-specific funding can be difficult to secure, but the researchers and managers had a "story: this marsh was unstable… it's basically already drowning. And so that was enough to convince all these funders to raise millions of dollars to test it."[23] Second, the marsh was in such poor condition that "they had nothing to lose."[24] Prior to sediment augmentation, "the entire marsh, including all the cord grass, [was] completely submerged at the high tide."[25] Conventional habitat restoration and plant control practices were irrelevant because "by the next fifty years, we're going to be under water."[26] The science-management team were presented with an opportunity to experiment with conservation actions that allowed them "to put dredged material on top of endangered species habitat."[27] In this case, the poor conservation prospects of the marsh gave latitude to bold experimentation, which acted as a catalyst for multi-actor involvement and funding.

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[21] Interview 13
[22] Interview 19
[23] Interview 19
[24] Interview 19
[25] Interview 13
[26] Interview 13
[27] Interview 19

Third, extensive time and resources were devoted to the project – both a contributor to and consequence of success. This involved not only deliberate coproduction (Meadow et al., 2015), but "coproduction to the magnified degree"[28] with "a lot of open communication."[29] This includes monthly phone calls with managers, scientists and funders, updates from the refuge, and sharing data and graduate research assistants across the various research and university teams. There is "huge visibility and accountability for this project."[30] But, as another science-management actor notes:

> If…, the real way to make a difference in terms of resource management is to have these embedded teams where you're putting scientists with resource management, defining the problems, doing all this boundary science … then it's a remarkably small number of problems that the scientific community can actually address. Because there's not enough scientists to do this everywhere, with everybody and all the problems.[31]

Although the case of sediment augmentation in the coastal marsh suggests the importance of engaged, available teams, it is not clear that these opportunities and this amount of effort are scalable globally, due to environmental differences. However, this process might be scalable to other wetlands in California.

A shortcoming of this example is that it was a "failure" in adaptation strategy. However, what transpired was an unforeseen complication, not a failure. This complication was due to the physical parameters of the sediment, as there was a mismatch between the characteristics of the applied sediment and depth of sediment applied and the natural conditions of the marsh.

[28] Interview 19
[29] Interview 13
[30] Interview 19
[31] Interview 5

However, it is important to note that while the sedimentation application had complications that slowed the recovery of the marsh, the augmentation project in general was a success in terms of science coproduction. As one actor put it, “I would have never been involved in this study if the SW CASC had not brought the actors together… this collaboration greatly expanded my expertise in science coproduction efforts”[32]. The success of this project was its ability to bringing scientists, environmental planners, and managers together, providing the nexus for future coproduction efforts and working together on more potential techniques and adaptation strategies. Additionally, this project is an example of successful adaptive management. The monitoring 6 months before and 5 years after the sediment application showed there were problems with vegetation establishment, so new efforts were made to establish plants (by planting them, as an experiment). These adaptive responses to challenges that arose during the project were made possible due to the collaborative nature of the work, which enhanced the overall success of the augmentation project.

Notwithstanding many examples of coproduction, boundary organizations, and climate services, there are few instances where such collaboration is seen to actually inform decisions. Even those with careers dedicated to coproducing actionable science have difficulty identifying examples where their science has influenced management: “I can’t point to anything that I’ve done that’s had those kinds of effects.”[33] The reasons for this, as nominated by interviewees, echo insights from the long-standing science-policy literature. Information provided by scientists is not actionable in management decisions: it is “at the wrong scale, too big a scale, … it’s not applicable to their day-to-day work.”[34] Some boundary organizations continue to produce

[32] Interview 31
[33] Interview 11
[34] Interview 2

climate downscaling or models with the assumption that "better interpretations of the consequences of climate change to ecological systems [means that] resource managers will know what to do. … [but] that's rarely the case."[35] For the "folks that are … touching the species, walking through the habitat"[36] downscaled climate information is not always immediately usable. Similarly, there are issues with "timeliness", where "the question you may have had, like, three years ago, is no longer relevant."[37] Institutional barriers can potentially undermine collaboration, including overhead fees charged by USGS or universities, and the complex contracts that these collaborations require.[38] However, we have shown how institutional assistance, such as the SW CASC and university workshop collaboration, highlight a way forward. Bridging coproduction and climate services, by, in a sense, creating the network, brings together these separate entities.

Perhaps most importantly, and underlying the above comments, access to useful climate services does not always facilitate a clear management decision. One interviewee describes: "if you were to do some very nice science that says that a particular fish is in deep, deep trouble, it's going extinct. And your evidence is rock-solid. … It's *this* species is going extinct in *this* location. That doesn't necessarily mean that the resource manager has something to do, an action to take."[39] On the one hand, the resource manager is constrained by their decision-making context. As one scientist notes, such decisions are "a different political kind of process" than making actionable science: when "you asked me to look at the science on X, Y, and Z, and I did. And now what you do with that, … that's up to you."[40] On the other hand, the 'what to do'

[35] Interview 5
[36] Interview 14
[37] Interview 4
[38] Interview 3
[39] Interview 5
[40] Interview 11

question goes to the heart of uncertainties in conservation science more broadly. One state Fish and Wildlife agent describes how their model of acquiring land for species habitat – a pillar of their conservation strategy – is now up in the air: "We spent several decades in the department acquiring land for migratory deer… [land] where they would fawn, and other areas where they would over-winter. Those areas are really kind of useless now for the species, you know, they're not showing up there anymore. Where are they showing up? We haven't acquired those lands yet. … Going out and acquiring lands… that model's done."[41] Climate change will continue to alter the coastal landscape, and new methods will have to be adapted to better support resource managers and policy makers in their conservation efforts.

Learning from the experiences of the first five years of the SW CASC, the experience of scientists working together on the Seal Beach project, and self-reflection on behalf of the SW CASC scientists, two different models were proposed to address the integration of science to policy. First, creating more local and intimate spaces to attract the full range of stakeholders with the goal of providing networking opportunities and room for discussion that can lead to science coproduction. Second, recognizing the importance of early-stage education of the next generation of scientists, the SW CASC implemented the Natural Resources Workforce Development (NRWD) Fellowship.

The first model was a current attempt at creating more local and useful networks through a collaboration between the SW CASC and UCLA. This included scientists from UCLA who had been involved in the Seal Beach coproduction effort. The approach was also informed by their experience in early SW CASC workshops where scientists talked to managers about climate

[41] Interview 15

change, but no coproduction activities resulted. Based on the first phase of participant observation study and interviews, it became clear to the UCLA scientists that the dialogue had to start the other way around – with planners and managers telling scientists about the climate challenges and information needs that existed in their management domains. No single government or community group can tackle the large questions that arise from climate change alone. California's history of success and replication in environmental policy provides an experimental space to engage multiple actors collaboratively to increase possibilities for the circulation of new ideas and strategies. Thus, the intent of this project was to bring together coastal reserve and resources managers and planners to share their current priorities with each other and climate change scientists, and facilitate sharing of information between participants, to help determine the research and information products that the SW CASC should prioritize to assist managers and planners, and to inspire science coproduction collaborations. In order to set up a framework and explore replication, a series of workshops were created that expanded on prioritized topics that arose from the first workshop.

The potential success of this endeavor was apparent, as one participant notes "I've been around long enough to see a change in management may have different values…our programs are constantly getting waddled down in terms of funding and staffing…at the park service, the funding is not in perpetuity, that is why I am participating, this is such a fantastic group…you guys know the issue, but I see it being extremely important to follow up on some of these big issues, to say more than just the trends…" (SW CASC UCLA Coastal Managers Workshop #2, 2022). It is important to note that, similar to the previous success story, most participant members were seasoned and familiar with multi-institutional work. Higher rates of collaboration and coproduction can be achieved when networks of people have established rapport, since participants can be more “comfortable dealing with those issues because [they’ve] all worked with managers” (SW CASC UCLA Coastal Managers Workshop #2, 2022). At this point a group of the participants, including managers and scientists, are working on developing joint research proposals.

One criticism of this effort is that it can be hard to replicate. Firstly, these workshops are established through specific funds that highlight the necessity of science coproduction efforts. Additionally, these networks take a lot of time to establish, with big upfront time-investments. Similarly, once these networks are established, they need to be upkept and maintained over long periods of time. While groups of participants may have “more research questions than we have time or bandwidth” (SW CASC UCLA Coastal Managers Workshop #2, 2022), unless there is an immediate call for proposals, they often move more slowly in terms of funding coproduction projects. One way to address this is with incentives and resources. At the current moment, the coastal coproduction network is still trying to develop the resources. Having access to the network, as well as readily prepared collaborative projects for when funding opportunities arise, provides a leg-up for getting coproduction focused funds. For future efforts, there needs to be more programs, like the CASC, that demand coproduction as part of the product in their calls for proposals. A recent example of this on the national scale includes the 2022 Actionable Science Funding Competition as part of NOAA’s Restore Science Program.

As a component of its efforts to enhance the climate science – policy interface the SW CASC instituted the NRWD program to provide graduate students from seven consortium universities with opportunities for training and practice in developing use-inspired and actionable science to inform natural resource management decisions. Graduate students are trained in methods to foster collaborations and the development of science that informs resource management decisions, interacting and collaborating with natural resource management decision-makers, and getting experience in effective communication of research results to enable use of that research (https://www.swcasc.arizona.edu/nrwd-fellowship). One of the researchers in this study is a fellow of the first-year cohort and highlights the importance of training in science-policy and coproduction integration during the early years of their academic career. This fellowship elaborates on the themes of science coproduction, expanding upon the methods that can be utilized in research and management settings. Graduate students received in-person training in translational ecology and interdisciplinary collaboration, with multi-day intensive lessons, and engagement in role-playing activities. This experience as an early career scientist informed future participation in the coastal workshops that the SW CASC and UCLA had put together. Additionally, the fellowship not only taught the fundamentals of aiming and organizing a workshop, but how to effectively communicate with practitioners and understand their differing priorities. The themes of science coproduction reverberated throughout the multi-year series due to the training and guidance in effective integration of climate science and climate adaptation science into management and planning of environmental resources.

In summary, the science-policy models of coproduction, boundary organizations and climate services have been applied in numerous cases to encourage climate smart conservation and resource management decisions in California. Yet there are few instances where useful science informs management decisions, despite extended investments. The experimental coastal marsh sediment augmentation project is promising in this respect, but required substantial financial and human resources, suggesting limits to replication and scale. The coastal workshop environment provides an avenue forward, but may present geographical limitations, as will be expanded upon. Education of the next generation of climate scientists in the importance and development of coproduction and science-policy integration, through initiatives such as the NRWD Fellowship program of the SW CASC, are an important potential contributor to increasing capacity, but have thus far only been able to see limited deployment.

4.7 Discussion

4.7.1 Challenges of Current Models

In section 4, interviewees suggest that climate change has triggered a proliferation of science-policy collaborations in the resource management arena. This is certainly true, but these science-policy/management experiments also emerge within and can be constrained by specific decision-making culture and political economic context. Models of funding embedded in the current economic system are important here.

Several actors noted the power of funding to convene researchers and decision-makers, but also influencing scientists' questions and methodologies. One challenge highlighted by interviewees is how this funding often requires tangible results, such as measurable indicators and action items deliverables. As one funder of applied research notes (and others echoed): "we do get a lot of pressure from our legislature to say, you know, we're using… hundreds of millions of dollars on research, what are we getting out of it?…how is this benefitting my constituents?"[42] Yet the immediate efficacy of coproduction and boundary organizations is difficult to measure: when "a steering committee comes together and meets for two days in June, what does that result in on the ground for conservation, within a year, or five years? … Sitting around a table having conversations, how is that paying off?"[43] Facing an audit culture of

[42] Interview 4
[43] Interview 26

increased evaluation, accountability and surveillance, scientists seeking to create useful knowledge now must also attend to innovative measures that satisfy their political patrons. That is why it is important to have institutions who specifically provide funding for coproduction efforts. Funding institutions must highlight the importance of these efforts and support them fiscally to provide creative freedom and support in climate science and climate adaptation science research. While overhead fees can be sometimes be costly, conversely, institutions can provide financial services to support collaborative projects. Boundary institutions and organizations like the CASCs, which are funded by DOI through USGS, receive substantial funds that support bringing together diverse groups to do coproduction research. Whereas the current model supports agencies putting out calls for proposals, this new model funds bottom-up science coproduction efforts that look for resources based on an idea, supporting new synergies and productive collaborations. Additionally, these larger institutions provide a way forward for issues in scalability, and replicability, of research projects like Seal Beach, by having access to and providing the resources necessary to bring together multiscale collaborations to tackle large-scale climate change problems. This is due to the CASCs acting not as private institutions, but as government funded institutions. While promising, these efforts are relatively recent and ambitious in their aims and require further study.

Public institutions are also underwriting new science-policy models that reflect emphases on localization, outsourcing, and privatization of knowledge production. One state agency employee explains: science-policy experiments must "confront… the civil service system."[44] For decades, there has been an emphasis on "contracting out [to independent institutions] for more government work because it was deemed to be cheaper" than hiring a continuing government

[44] Interview 4

employee. Rather than employing researchers alongside resource managers within state institutions, scientific research is contracted to universities, consultants, and NGOs. As this interviewee noted, aside from a final report or assessment the agency loses access to its ongoing use and application. Similarly, NOAA has recently embarked on a phase to facilitate a growing role for private consultants as providers of climate services, seeking more responsive and faster information for adaptation efforts. As an employee explains, this is because NOAA is currently too 'clunky' and "climate change … is a problem for these stiff institutions."[45] An increasing distaste for State and Federal governments as providers of public knowledge also manifests in a devolution of responsibility to more localized decision makers – for "flexibility… to achieve [adaptation]."[46] Underlying these changes, and reflecting an institutionalized desire for shifts towards supposed efficiency through contracting and devolution, is the now prevalent assumption that public services are overly burdensome and bureaucratic. Indeed, this "public perception of civil service"[47] is reported as one of the biggest challenges of the Federal Climate Science Centers.

Yet, it requires considerable institutional resources to manage science-policy interactions as they are privatized and localized. One challenge for boundary organizations is to avoid including too many actors that produce ever-longer boundary chains, resulting in confusion for those seeking to contact and establish networks with the proper personnel. Be it a state agency, local government, or non-governmental organization, or a research or scientific institution, "there is a whole other sub-ecosystem of how that whole chain works, going from the technical staff …

[45] Interview 24
[46] Interview 3
[47] Interview 28
[51] Interview 22

to the people who ultimately make a policy or a decision."[48] A water manager describes: "when something is very diffuse and there are many, many players, it's very difficult to know where do you begin."[49] Additionally, the proliferating number of actors and institutions, and increased distance between scientists and managers, can also have perverse effects on the science itself. A long-term observer indicates that there are more and more "question[s] of … quality control as the information moves back and forth"[50] between producers, consumers, and various intermediaries. In the process, scientists note that "some of your products get used by people that are not directly connected."[51] This should be celebrated on occasion but can also result in scientific information being used in ways that stretch its intended application. This is especially delicate when working with climate data, where uncertainty plays a role in future projections and needs to be understood to be interpreted correctly.

There are many challenges that face coproduction efforts, boundary organizations, and the distribution and use of climate services. Additionally, these processes and their effects unfold in uneven and geographically contingent ways. Often, these processes are looked at independently, as closed systems, rather than collectively. And while coproduction, boundary organizations, and climate services have their unique strengths, bridging them together within a network can have positive outcomes, as these entities don't necessarily have to work independently and are stronger together.

Coproduction efforts produce high-quality knowledge when there are a diverse set of actors that work in context-based, pluralistic, goal-oriented and integrative settings (Nordström

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[49] Interview 22
[50] Interview 22
[51] Interview 27

et al., 2020). We have seen how, through funding aimed at actionable science, the Seal Beach NWR project brought a diverse set of scientists, coastal land managers, and other partners to work collaboratively to restore a marsh. Through their adaptive-management process, ongoing open communication, and multi-institutional support, they were able to produce research that educated and inspired the actors involved in science coproduction, providing the nexus for future coproduction efforts. Additionally, this project was created with the idea to scale up from Seal Beach NWR without having to replicate all the research and detailed monitoring that was required. Indeed, a large effort like this for a particular management problem doesn't have to be repeated everywhere; the lessons from it can be applied without doing new studies. In this sense, coproduction skills are learnable and transferable, aiding in future projects in climate adaptation strategies.

Boundary organizations have the ability to create flexible management structures within the resource management agencies that aid in practitioners' ability to collaborate with research teams (Djenontin and Meadow 2018). Additionally, similar to the Seal Beach project and the NRWD fellowship, they provide the funds to support more coproduction work. The importance of the availability of coproduction funding opportunities cannot be understated. Boundary organizations are crucial to highlighting the importance of science coproduction efforts, as well as educating the future of climate adaptation scientists in addressing sustainability challenges.

Brasseaur and Gallardo (2016) suggest that climate services and climate service providers "should host under the same roof a diversity of specialists as well as stakeholders representing the corporate world and public services as well as social managers, engineers, and communication specialists". As interviewees highlighted, when information such as data or tools for research are moved back and forth between various actors, scientific information can end up being used in ways that have unintended consequences. Having a diverse set of specialists, stakeholders, scientists, and other actors under the same roof provides ease of access to those that can aid in supporting and educating users of climate services.

The CASC institutions attempts to bridge these independent processes by creating localized networks that bring together managers and scientists to keep in touch in more informal and intimate settings. There are geographical regions set up by the CASC, which strengthens the connections more locally, while still providing an overhead institution that is directly connected in a national capacity. The networks create spaces where people can work together, and have an ease of access to one another, allowing for interactions and collaborations with different parties from different organizations. The boundary chains are shortened as partnerships are built and strengthened on trust and communication, and the potential for real coproduction efforts can take place in meaningful ways. As one coastal manager suggests: “I would rather focus on the strengths of this group rather than…hire someone to do that work…It’s more responsive to have the academics go after pots of money, and the feds can’t actually go after certain pots of money…there are probably some really tangible things that we would want answered that would be complimentary to our existing reports and management plans that we can’t do but we know a group like this can, and see if there is a will within this group to submit funding to address these questions, that is the magic potion” (SW CASC UCLA Coastal Managers Workshop #2, 2022). Additionally, climate services are established, produced, and shared with more intention and efficacy due to on-going communication between actors in established coproduction efforts. This diminishes the ability for data or tools to be used incorrectly and creates more room for communication and education of shared information.

4.7.2 Regionalism and Climate Science – Policy Potential

Climate knowledge priorities are deeply connected to the political, fiscal, and regulatory landscape. Beyond globalizing trends related to the commercialization of knowledge that matter for science-policy, specific regional processes also shape locally produced science. In the case of California, regional factors include state policies and state identities, imaginaries, and norms. Assembly Bill 32, California's Global Warming Solutions Act of 2006 (AB-32), provides funding for research about carbon sequestration through its Greenhouse Gas Reduction Fund, thereby shifting research and management practices. As one NGO scientist admits, even though "most of the [conservation] biologists here, we're not trained to think about carbon,"[52] research abounds about meeting conservation and sequestration goals. While encouraging research about sequestering greenhouse gases is precisely the purpose of this funding, there is the danger that the need for research grants detracts effort from other important areas of research that would inform natural resources management.

In addition to regulatory influences, the science-policy interface reproduces and is enabled by particular Californian imaginaries and identities. As one interviewee noted, the expanding landscape of science-policy efforts is imbued with the "rugged individualism, and the whole mythology of the West."[53] Even with efforts that are Federal in scope, Californian interlocutors invoke environmental, social, and political histories to explain their efforts. As several observers explained: "Of course, as you know, the international community looks to California as a leader to address [climate], energy, and environmental problems,"[54] a state that is

[52] Interview 17; also Interview 23
[53] Interview 15
[54] Interview 4

"an innovator."[55] They describe California as having really "moved on climate change, and they have a culture of responding to disasters."[56] Investments across the science-policy interface are framed as an environmental necessity. "California has been on the leading edge of trying to understand and prepare for climate change" because it is so dependent on its diverse environments.[57] Californian economic and social endowments are also important, report the interviewees. California is "unique in a lot of ways" including in size, economically, and culturally.[58] As "the eighth largest economy, most populous state in the Union, a very environmentally conscientious and well-educated population; … those factors have really contributed to both our leadership and success at trying to address climate change in the state."[59] These factors are described as transcending political ideologies: "two very different administrations… [Schwarzenegger and Brown] both made the calculation that climate change is an extremely important issue for the state"[60] and have responded accordingly. Citing economic, political, and social endowments to explain climate change actions belies a much more contradictory climatological and environmental history, however. The imaginary of California as a climate and environment steward can potentially cover over conflicts within the state, in particular about water resources and land, and the ecological relationship with neighboring areas.

This imaginary also hides that the ecosystems spawning science-policy collaborations have done so because "there's this sense of urgency, [with] so many ecosystem threats for so long."[61] Yet, Méndez (2020) argues that this sense of urgency is influential in the success of

[55] Interview 24
[56] Interview 24
[57] Interview 7
[58] Interview 8
[59] Interview 4
[60] Interview 7
[61] Interview 19

California’s leaderships in climate change adaptation and mitigation efforts. California’s leadership is, in part, due to the collaboration between different states and global provinces, as well as more local community residents, as seen with the AB-32 bill (Méndez, 2020). These local and more global communities have worked together to bypass federal inaction, which results in real changes in environmental policy. California’s leadership on climate change demonstrates the ability of subnational governments to link local concerns with global forums (Méndez, 2020).

Regionalism in the context of mitigation and adaptation efforts in conservation science is necessary due to the different spatial and temporal scales that polices are defined by. Large climate change issues require local managers and planners to produce localized data for these efforts. This is due to the geographical differences in adaptation and mitigation efforts. Adaptation benefits (ways to prepare society for climate change impacts) are local and short-term, while mitigation benefits (which deal with the causes of climate change) are global and longer term (Méndez, 2020). However, this is where the benefits of national-level efforts, like those of the CASC, show their value. These larger, connected institutions are trying to get past localization issues by having a multi-state organization that provides information as a government service. In a sense, the CASCs attempt to provide national coverage, with multiple groups of institutions that are designed and supported evenly, so that if a state has limited funds for climate data and research, this national entity attempts to override local limitations to support localized research through access to regional networks and climate services. Additionally, these multiple groups of institutions are made to fit each geographical and climatic region, which allows them to address and support more localized needs. What the CASC model provides is essentially a distributed network where different states (and their university counterparts) have local, regional, and federal representation, as well as fiscal access that is sensitive to the needs of each state.

4.7.1 Educating the Next Generation of Climate Scientists

The NRWD fellowship program of the SW CASC was established with the aim of encouraging greater impact and promoting multidisciplinary and interdisciplinary research among the natural and social sciences. Specifically, their aim is to educate and develop the next generation of climate science and climate adaptation science researchers at the earliest stages of their careers. They are not alone in their efforts, as Eakin and Pratt (2011) suggest, “global, regional, national, and subnational scales, a variety of governmental and nongovernmental actors are now engaged in adapting their activities to a changing climate, promoting others to engage in adaptation, working to build adaptive capacity, and negotiating who shall pay for it”. Being a recipient and trainee of established and well-funded institutions like the SW CASC provides an opportunity to reflect on the value and shortcomings of their initiatives, such as the NRWD fellowship program, and their subgrant collaborations with consortium universities.

One large benefit of the NRWD fellowship is their integration of education at the early stages of recipient’s careers. Often, recipients have little to no exposure of what climate adaptation research looks like in theory and in practice. The fellowship not only provides insight and guidance to the design and practice of adaptation research, but they also provide hands on training which allows recipients to get a more holistic picture of what scenarios might arise in climate adaptation research. In particular, their focus on improving awareness of potential conflicting intentions present in adaptation projects, as well as training in the language and procedures necessary for conflict resolution in these environments is a strength in the program.

This comes through not only in the training before the project begins, but in the project itself, as invariably priorities shift and collaboration within a group of participants with diverse backgrounds can expose differing ideals. In this capacity, the training and program are very well catered to provide the recipients with the education and tools necessary to apply their knowledge in future collaborations of climate adaptation research.

Additionally, a significant benefit of this fellowship training program is that it addresses the issue of a lack of recognition of boundary spanners as a distinct profession (Goodrich et al. 2020). Due to this lack of recognition, there is a lack of training and evaluation of the researchers and managers who engage in boundary spanning activities. However, the NRWD training addresses this issue by allowing for the next generation of scientists interested in climate change adaptation research to train in the profession of boundary spanning, learning from more experienced members of the community already engaged in active boundary spanning work, and expanding the network which allows for more robust assessments of these efforts by the community.

The fellowship emphasizes collaboration of diverse groups by bringing together and utilizing recipients from different academic backgrounds. They highlight the value in creating networking spaces that nurture the establishment and maintenance of relationships with collaborators. This education expanded beyond the fellowship and informed the work that was done in collaboration between the SW CASC and UCLA. The coastal workshops were created with similar ideals, focused on creating on-going engagement, in the form of symposium meetings, collaborative reports, and mentorship meetings were created by the project collaborators to facilitate science coproduction between the university scientists and the reserve managers and community leaders. This perspective shift allowed for the network to be managed as an ongoing collaborative space focused on relationship building, rather than product output and project deadlines.

With the above in mind, there are ways in which efforts led by the SW CASC and others can be improved. One of the challenges of climate adaptation science and resilience research in general is that it faces novel challenges that are often large-scale, timely, costly, and with a degree of uncertainty. These efforts require an understanding of multidisciplinary research, which draw upon a breadth of knowledge, thus requiring a diverse set of actors. While the SW CASC funds adaptation research and collaboration, is it not a requirement to set aside some of the funds to include someone trained in adaptation theory that can provide guidance to these initiatives. This can be someone trained as a boundary spanner, or other similarly trained position, that aid in the design and implementation of adaptation strategy research while incorporating the complex political, socio-ecological, and institutional components of field.

Additionally, while there has been an increase in research looking at the successes and failures of climate adaptation science research projects, there is still a lack of similar theoretical investigation and examination of existing adaptation institutions and partnerships (Jones et al., 2018; Cochrane et al., 2017). Most of the CASC networks were founded in 2008, with additional regional networks established in 2011 and 2014. With over a decade of institutional climate adaptation efforts extant, it is important to understand if and how the institutions themselves have adapted to evolutions in climate adaptation program design, by implementing internal or external evaluations of best practices. However, research suggests that these first-generation institutions have been slow to adapt, not keeping pace with changes in the approaches to research (Klein et al., 2017; Jones et al., 2018).

4.8 Conclusion

In California there are multiplying attempts to put coproduction, boundary organizations, and climate services to use for shaping adaptive responses to climate change through new science-policy models; however, our research found few examples of practices that are actually translated into effective management practices or other adaptation investments. Our research also unveiled unintended consequences inherent to these new models of science-policy interaction, reflecting the complex contexts from which they emerge and that they reproduce. We highlight the importance of a proper education in these models in order to embrace them effectively. Looking forward, we show how viewing different coproduction efforts as a collective, rather than closed systems, can have positive outcomes. However, we highlight the necessity of fiscal support from funding institutions to increase these efforts. While there are boundary organizations that create incentives for science co-production, universities also have an opportunity to increase incentives, not only by providing financial support, but by increasing recognition of these efforts within the research community.

We analyzed the broader geographical and institutional context in which these models arise and are put to work in Californian policy-making and resource management. We show that these models of science-policy interaction are shaped by their regional specificity and also habitually rely on and reproduce imaginaries of ecological stewardship. We highlight that coproduction efforts from boundary organizations, such as the SW CASC network, try to minimize the regional variability from place to place, making it easier to rely on best practices and lessons learned from other sites and contexts. Additionally, for institutions like the CASCs, we show how presence at the national and local scale allows for high level support, both creatively and fiscally, while allowing for there to be nuance in addressing the various needs of each locality which has unique needs in adapting and responding to climate change.

Nonetheless, this article does not seek to criticize the work of fostering different forms and processes for producing science that might inform the vital task of developing adaptation options and practices. Efforts across the science-policy network seek the kinds of diverse, abundant forms of life so threatened by climate change, with limited resources. Rather, we wish to highlight the importance of various forms of experimentation, like the ones highlighted in this paper, that are necessary to engage various groups of actors to collaborative in coproduction efforts. Creating intimate, localized networks of communities that have access to larger, national networks empower groups of local, regional, and national scale to tackle future climate change challenges.

4.9 Appendix

Appendix Table 4.1 Summary of institutions studied for this research project

Name	**Description**	**Examples**	**Interviews**
	Science producers		
Climate Assessment for the Southwest (CLIMAS) program; formerly *Regional Integrated Sciences and Assessment program (RISA)*	Overseen by US National Oceanic and Atmospheric Administration (NOAA); includes academics and agency researchers; Federally funded, regionally distributed	California-Nevada Applications Program (CNAP)	5
Climate Adaptation Science Centers (CASCs)	Overseen by the Department of Interior (DoI), through United States Geological Survey (USGS); includes academics and agency researchers; Federally funded, regionally distributed	Southwest Climate Science Center (SWCASC)	8
National Oceanic and Atmospheric Administration (NOAA)	Federal agency; agency researchers; various regional offices	Staff working in climate and societal interactions	2
	Intermediaries		
Landscape Conservation Cooperative (LCC)	Overseen by US Fish and Wildlife Service (FWS); agency researchers and managers; Federally funded, regionally distributed	California LCC	2
California Energy Commission (CEC)	Commission within California Natural Resources Agency; produces California Climate Assessment		2
Conservation NGOs	Intermediaries and consumers (as land managers)	Point Blue Conservation Science; The Nature Conservancy	4
Consultants	Private companies	Take on a great variety of roles in different projects	1
	Science users		
California Natural Resources Agency	State agency		1
Department of Water Resources	State department within California Natural Resources Agency		2
Department of Fish and Wildlife	State department within California Natural Resources Agency		2
Fish and Wildlife Services	Federal agency within US DoI		3

Appendix Table 4.2 List of interviews

Interview 1	California Energy Commission, 22 February 2016, Sacramento CA
Interview 2	Landscape Conservation Cooperative, 22 February 2016, Sacramento CA USA
Interview 3	Natural Resources Agency, 23 February 2016, Sacramento CA USA
Interview 4	California Energy Commission, 23 February 2016, Sacramento CA USA
Interview 5	University of California, Davis, 24 February 2016, Davis CA USA
Interview 6	University of California, Davis, 24 February 2016, Davis CA USA
Interview 7	University of California, San Diego, 17 March 2016, La Jolla CA USA
Interview 8	University of California, San Diego, 17 March 2016, La Jolla CA USA
Interview 9	University of California, San Diego, 17 March 2016, La Jolla CA USA
Interview 10	University of California, San Diego, 17 March 2016, La Jolla CA USA
Interview 11	University of California, Davis, 1 June 2016, Davis CA USA
Interview 12	Fish and Wildlife Service, 2 June 2016, Chula Vista CA USA
Interview 13	Fish and Wildlife Service, 2 June 2016, Seal Beach CA USA
Interview 14	California Department of Fish and Wildlife, 8 June 2016, Sacramento CA USA
Interview 15	California Department of Water Resources, 8 June 2016, Sacramento CA USA
Interview 16	California Department of Fish and Wildlife, 9 June 2016, Sacramento CA USA
Interview 17	Point Blue Conservation Science, 10 June 2016, Petaluma CA USA
Interview 18	The Nature Conservancy, 13 June 2016, Sacramento CA USA
Interview 19	United States Geological Survey, 14 June 2016, Vallejo CA USA
Interview 20	Point Blue Conservation Science, 21 June 2016, Petaluma CA USA
Interview 21	RAND Corporation, 22 June 2016, Santa Monica CA USA
Interview 22	Private Consultant, 8 July 2016, Santa Cruz CA USA
Interview 23	The Nature Conservancy, 12 July 2016, Sacramento CA USA
Interview 24	NOAA Climate Program Office, 22 September 2016, Silver Spring MD USA
Interview 25	NOAA Climate Program Office, 22 September 2016, Silver Spring MD USA
Interview 26	Fish and Wildlife Service, 23 September 2016, Falls Church VA USA
Interview 27	United States Geological Survey, 23 September 2016, Reston VA USA
Interview 28	United States Geological Survey, 23 September 2016, Reston VA USA
Interview 29	Association of Fish and Wildlife Agencies Offices, 26 September 2016, Washington DC USA
Interview 30	NOAA Climate Program Office, 22 September 2016, Silver Spring MD USA
Interview 31	University of California, Los Angeles, 8 August 2022, Los Angeles CA USA

4.10 References

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5. Conclusions

Currently, nearly 2.4 billion people live within 60 miles of the coast, more than 600 million live in low elevation coastal zones, and 150 million people live within 1 m of high tide (Neumann et al., 2015; Jevrejeva et al., 2014; Lichter et al., 2011; McGranahan et al., 2007). This human proximity to coastal environments adds additional stress to an already heavily used and threatened natural system (Barbier 2019). Centuries of anthropogenic impacts from industrialization have altered sedimentation regimes and contributed to heavy metal pollution in coastal habitats (Barnard et al., 2013, Gehrke et al., 2011; Luoma et al., 1998). Although local land-use and pollutants are important stressors on California's coastal environments, sea level rise (SLR), caused predominantly by global rising temperatures, will be one of the most damaging factors to coastal communities in the 21st century (Anthoff et al., 2009). Recent model estimates predict that sea levels will rise from 16-61 cm by 2050, and up to 2 m by 2100 (Bamber et al., 2019). Consequently, there will be increases in frequency and intensity of storm surges and coastal flooding (Teegavarapu and Schmidt 2019); monsoons and cyclones (Schewe et al., 2011; Gupta et al., 2019); heat waves and heavy precipitation (IPCC 2022); Arctic and Antarctic melting (DeConto and Pollard 2016; Serreze and Meier 2019); and coastal erosion and population displacement (Vitousek et al., 2017; Neumann et al., 2015).

Historical observations show that tidal wetlands can tolerate high levels of SLR, however growing concentrations of atmospheric CO^2, increased temperatures and thermal warming of the oceans, as well as population growth and coastal squeeze bring in unprecedented factors that challenge efforts to understand future salt marsh trajectories (IPCC 2022). With projected increases in climate change impacts on the horizon, it will be crucial to look at these systems as a whole by incorporating spatial variables and localized biophysical feedback processes in future marsh vulnerability assessments and modelling efforts. Additionally, while there is some level of regional disparity, human activity can have major impacts on ecogeomorphic feedback systems and shifting thresholds across the marsh complex. Therefore, anthropogenic stressors should be coupled with climate change impacts in management and conservation efforts, as they often interact synergistically.

The work for this PhD dissertation aimed to understand the multiple stressors on California coastal marshes. Sedimentological records of heavy metal contaminations have highlighted the impacts of anthropogenic pollution levels in salt marsh sediments, which are of major concern to the disruption of biodiversity and ecosystem resilience (Córdova-Kreylos et al., 2006; Zhang et al., 2013; Bhuyan et al., 2017). Additionally, understanding long-term spatial and temporal changes in SLR and its impacts is crucial for creating policies and implementing management practices that protect coastal habitats and communities (Wasson et al., 2019). To inform these conservation and management decisions, robust assessments of habitat and ecosystem dynamics trajectories are needed. In the second chapter, I utilized the longest and highest resolution heavy metal accumulation data from three San Francisco Bay marshes to examine European impacts in the context of earlier Holocene variability. Results confirmed significant European impacts on the geochemistry of marshes in the San Francisco Bay over the last ~150-200 years, while highlighting that post-European concentrations are not so far removed compared to pre-European maximum concentrations. Additionally, over the past few decades, the trajectory of a number of indicators, including notably Pb, are returning to conditions more typical of the immediate pre-European period. However, although these trajectories suggest resilience and return to more natural ecological conditions, how anticipated 21st century sea level rise will impact these marshes remains an important concern.

In the third chapter, I tackled the issue of nature-based engineering interventions to mitigate SLR in coastal marshes. I analyzed sediment cores from the Seal Beach NWR, to determine the nature of the pre-existing sediments of the site and to understand natural accretion rates and variability over time, comparing results to the augmentation sediments and depth of sediment applied during a 2016 experimental augmentation treatment. Results revealed that, although the cores revealed natural variations in the grainsize and organic content of sediments deposited at the site over the past 1500 years, the applied sediments were markedly coarser in grainsize than prehistoric sediments at the site. This mismatch strongly contributed to the slow recovery of marsh vegetation observed at the site in the five years following the augmentation experiment. Generally, within the temperate zone, ideal conditions such as gradual tidal prism regimes, reliable sediment sources, and accommodation space availability promote salt marsh resiliency. While progress has been made with regards to these processes individually, more work is needed to incorporate these individual elements into multidimensional frameworks to inform future management decisions and adaptation strategies.

In tandem to understand the multiple stressors on California coastal marshes, this PhD dissertation aimed to understand how the knowledge produced from traditional scientific research becomes woven into the fabric of management and policy settings. In the final chapter, I have argued that, in California, while there are multiplying attempts to put coproduction, boundary organizations, and climate services to use for shaping adaptive responses to climate change through new science-policy models, few are actually translated into effective management practices or other adaptation investments. Additionally, there can be unintended consequences inherent to these new models of science-policy interaction, reflecting the complex contexts from which they emerge and that they reproduce. However, I highlight how viewing different co-production efforts as a collective, rather than closed systems, can have positive outcomes, while also highlighting the necessity of fiscal support from funding institutions to increase these efforts. Finally, I show how co-production efforts from boundary organizations try to minimize the regional variability from place to place, making it easier to rely on best practices and lessons learned from other sites and contexts.

The work for this dissertation spans broadly, which has provided me with the opportunity to reflect on the interconnectivity of this research. As such, I have some recommendations for future environmental researchers, as well as educational institutions, who want to engage in and support science policy research and science co-production efforts. For researchers working on coastal and other environmental projects, I would recommend connecting with the variety of actors and stakeholders who engage in management of those lands at the beginning of a research project. This includes environmental managers, indigenous communities who use the land, and other stakeholders. Opening the dialogue outside of academia early on provides the opportunity to create meaningful relationships, improve management decisions, and enhance knowledge co-production efforts. Additionally, I would recommend the implementation of long-terms monitoring of projects in collaboration with managers, which would improve research efforts by supporting successful adaptive management.

With regards to educational institutions, such as UCLA and the SWCASC, there are some efforts that can be made to enrich the science policy overlap and inspire future researchers interested in science-coproduction efforts. First, I recommend that professors use the classroom as an opportunity to share and discuss the science co-production literature. Many students have little to no exposure of the breath and variety of science co-production efforts in environmental research. Next, I recommend that departments include researchers who engage in boundary spanning work in their colloquium speaker series. Highlighting their efforts provides insight into what they do for new generations and may show departmental and school-wide support for these efforts. This is also an important way to support academics who continue to engage in this type of work despite the lack of incentives and recognition provided. These are just a few recommendations that I hope can inspire new researchers and create possibilities for more inclusive research in the environmental sciences.

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