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# Understanding the relationship between sedimentation, vegetation and topography in the Tijuana River Estuary, San Diego, CA.

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UNIVERSITY OF SAN DIEGO

San Diego

# Understanding the relationship between sedimentation, vegetation and topography in the Tijuana River Estuary, San Diego, CA.

A thesis submitted in partial satisfaction of the
requirements for the degree of
Master of Science in Environmental and Ocean Sciences
by
Darbi R. Berry

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# 1. Chapter 1

## 1.1. INTRODUCTION

### *1.1.1. Coastal Wetland and Estuaries*

Coast lines provide space for a wide array of ecosystems, among these, coastal wetlands and estuaries are some of the most important. Estuaries are partially enclosed bodies of water that are formed at the interface of where a river meets the ocean. They are often classified by having both fresh water and salt water mixing, and their physical make-up can vary based on geographic region. Coastal estuaries and wetlands are uniquely located at the transitional interface between the aquatic and terrestrial ecosystems, which lends itself to their high productively both economically and environmentally. Barbier et al. 2011 indicates that the overall cumulative benefits these systems produce is greater than any other single ecosystem due to their unique location. Estuarine habitats provide both these economic and environmental benefits to the communities around them (Li et al. 2018, Kirwin and Megonigal 2013).

Environmentally, coastal estuaries serve as a natural barrier for storms and protection against coastal wave action (Barbier et al. 2011, Kirwin and Megonigal 2013). The research surrounding global climate change highlight an increase and severe storm activity. Estuaries provide a natural space for shoreline succession and retreat, which can help serve as a natural barrier for the communities surrounding them in the event of rising sea level (Kirwin and Megonigal 2013, Duong et al. 2015). Coastal estuaries are impacted by climate change driven variations in both oceanographic and terrestrial processes such as changes in rainfall and runoff.

Moreover, Doung et al. (2015) found that any negative changes are sure to result in socio-economic impacts Estuaries host an array of habitat zones, specifically coastal salt marshes. Many of these habitats serve as refuge for endangered and threatened species and as an important nesting ground for many local and migratory birds. The Tijuana River Estuary (TRE), in particular, is home to over 370 species of birds including the endangered Ridgeway Rail and the California Least Tern (Safran et al. 2017). Globally, coastal estuaries also serve as a nursery ground for an estimated 75% of commercially important fish species and 80-90% of recreational fish species (CERF). Costanza et al. estimated that the revenue generated annually by global wetlands at $33 trillion USD in 1997, and Barbier et al. (2011) later calculated that estuarine and coastal ecosystems provide $10,000 USD per hectare annually. For example, the total revenue from fish catch directly in estuaries in the United States was $4.3 billion USD per year alone (NOAA Estuaries Report 2012). Along with providing vital habitat space, estuaries serve as a sink and source for many pollutants. Their unique location, typically at the mouth of watersheds, allow them to operate as a holding space for much of the surrounding run-off. Both the marsh platform sediments and the plant species act as a natural sponge and filter out excess nutrients, pollutants and sediments (Breaux et al. 1995).

Estuarine habitats are found along coastlines around the globe, however due to their proximity to the coasts they have been among the most anthropogenically impacted ecosystems. Anthropogenic impact has caused increased degradation to many of the viable ecosystem services estuaries provide such as improved water quality and available habitat for critical and migratory species. One example of a valuable wetland ecosystem service is in southern Louisiana, USA, where Breaux et al. (1995) utilized marsh swamps for the treatment of wastewater. Due to their natural water filtration capabilities, the predominantly marsh swamps achieved a capitalized cost savings of $785 to $15,000 per acre (1 acre = 0.4 hectare) when compared to conventional municipal treatment for wastewater (Breaux et al. 1995). Globally, 50% of salt marsh habitats have been lost or degraded and have been greatly impacted by human activities such as dredging, draining, filling and damming (Barbier et al. 2011). In Southern California, in the last century estuaries have declined in area coverage over 90% (Zedler 1992). The estuaries that remain in southern California are subject to a long list of anthropogenic impacts due to drastic changes in both their upper and lower watersheds. One of the largest impacts to these systems is the increase in urban development (Biggs et al. 2015, Biggs et al. 2010, Taniguchi and Biggs 2015). This development has reduced the total area and fragmented of much of the remaining southern California estuaries, further decreasing the total natural space for retreat or habitat migration (Elwany 2011). The continued degradation of wetland ecosystems is certain to result in large socio-economic impacts, most of which would be felt closely by surrounding communities (Barbier et al. 2011, Duong et al. 2015).

Estuaries are often located at the mouth of a watershed system. These systems are not only impacted by degradation that may be occurring within or in close proximity to the areas surrounding the estuary itself, but also by changes occurring throughout the entire watershed. Increased urbanization has been shown to strongly influence natural watershed processes as a whole; such as the amount of sedimentation that occurs in response to seasonal rainfall events, in turn, the watershed's response to these changes lead by urbanization are dynamically altering some of the important salt marsh ecosystem functions (Biggs et al. 2015, Biggs et al. 2010, Taniguchi and Biggs 2015). Understanding both the coastal interface zone as well as upstream watershed processes is essential to holistic habitat management for estuarine systems.

## 1.2. Study Site

### *1.2.1. The Tijuana River Estuary*

The TRE is the southernmost estuary in San Diego County, located in the City of Imperial Beach, California. The estuary is located at the mouth of the Tijuana River, whose headwaters begin Southeast in the Laguna Mountains across the border in Mexico. Where the Tijuana River crosses the border, it creates the upper portion of the Tijuana River Valley, a freshwater, riparian, floodplain ecosystem dominated hydrologically by seasonal rain patterns. The study area us dominated by a Mediterranean climate regime, with warm dry summers and mild, cool wet winters. The main source of freshwater into the estuary is from the Tijuana River which has varying seasonal dependent flows (Elwany 2011). As the Tijuana River flows from East to West towards the mouth, the ecosystem begins to grade into the TRE. The Tijuana River is dominated by seasonal flows and outputs, which allow for saline and tidal conditions to dominate at the lower end of the system (Cahoon et al. 1996). The TRE is the largest intact estuary in Southern California and is home to a gradient of ecologically important coastal and tidal habitat zones. The TRE is a 1000 hectare coastal plain wetland that is tidally dominated at its mouth which is classified as a restricted or bar-built mouth. This classification indicates that mouth of the estuary may migrate laterally north and south along the coast as well as can close entirely under the right physical parameters (Zedler 1992, Cahoon et al. 1996). The intertidal wetland area has two distinct hydrological systems, the Northern Arm and the Southern Arm. Due to past and active restoration efforts, the Northern Arm has successfully seen a 97% restoration of its historical habitat types. Many of which are ecologically important and primary focus of this study including the Salt Flat (Pan) and Salt Marsh (SFEI 2017). This study focuses on the Southern hydrologic arm. The Southern arm has experienced much less success with restoration of its historical habitat types and is the subject of potential future restoration projects.

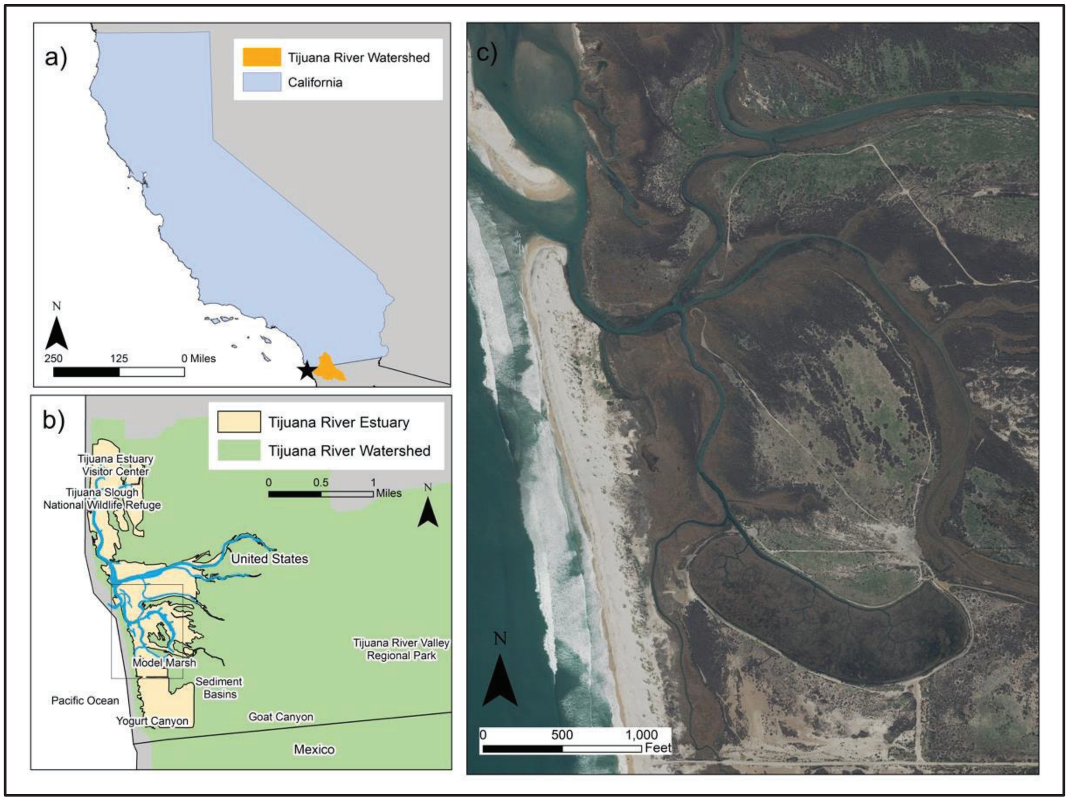

Figure 1.1. a) TRE is located in southern California and the mouth of the Tijuana River Watershed. b) The TRE is comprised of two distinct hydrologic units, the Northern Arm which includes the Tijuana Slough National Wildlife Refuge, and the Southern Arm which includes the Model Marsh, Sediment Basins, Goat and Yogurt Canyons. c) This research focuses on the Southern Arm of the TRE.

### *1.2.2. The Tijuana River Watershed*

The estuary is part of the Tijuana River Watershed, which covers an expanse of approximately 1,750 square miles; three-fourths of which lies across the US border in Mexico. Over the last half century many different water and water quality control policies have been implemented, both in the US and Mexico. The region is typically defined by the two large “sister cities” of San Diego in the US and Tijuana in Mexico. Over the last century, both of these locations have experienced large increases and growth in population therefore lead to an increase in urban development. This increased development in the upper Tijuana River Watershed may have influenced many of the environmental parameters that dominate the estuarine habitat. The two major parameters documented in literature driving restoration efforts are: the increase in sediment input into the estuary and an overall reduction in the estuary's tidal prism. Restoration projects here over the past two-decades have worked to address issues of sedimentation and habitat restoration. These projects recognize the large role tidal prism and availability play in habitat restoration and this research project hopes to provide supplemental data for current and ongoing restoration in the TRE.

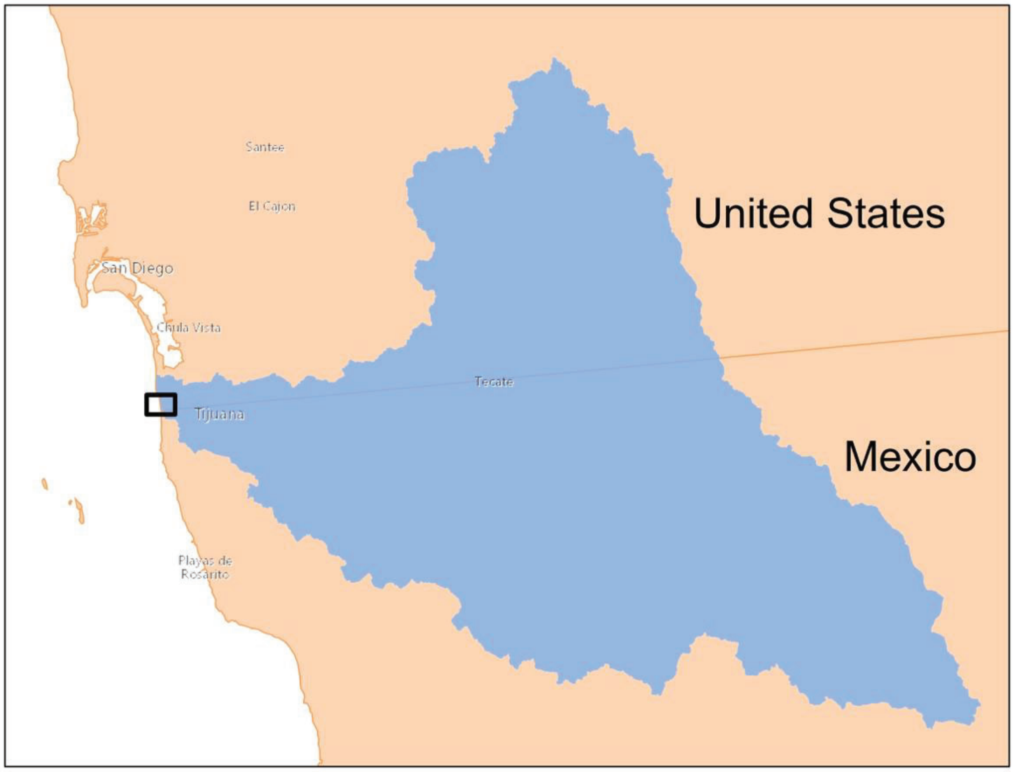

Figure 1.2. The Tijuana River Watershed covers 1,750 square miles and extends three-fourths of its area into Mexico. The TRE resides at its mouth.

## 1.3. Literature Review: Physical characteristics of interest

### *1.3.1. Sedimentation and Vegetation*

Salt marsh habitats are commonly defined by having a sharp zonation in plant communities as well as a low species diversity (Barbier et al. 2011). Largely the regulation of structure and function of salt marshes has been attributed to many physical processes such as elevation, salinity and flooding (Barbier et al. 2011, Mitsch and Gosselink 2008). The TRE in Southern California has long been subject to degradation and habitat type conversion due to increased anthropogenic activity which has led to an increase in the volume of sediment entering the estuary during storm and runoff events (Biggs et al. 2015, Biggs et al. 2010, Taniguchi and Biggs 2015 Wallace et al. 2005, Safran et al. 2017). Increased sedimentation has led to increased sediment accumulation on the marsh platform; overtime this aggrades, raising the marsh surface. This can lead to habitat type conversion throughout the marsh. Changes upstream can lead to conditions that may not be able to support the original habitat, and as a result habitat type conversion has occurred in the TRE (Safran et al. 2017). Wallace et al. (2005) defined fluvial sediment deposition as the primary management challenge in the TRE as well as many other southern California wetlands, precisely because the sediment deposition raises the marsh surface and fills creek networks which in turn reduces tidal connectivity (2005). The Tijuana River Valley Historical Ecological Investigation, a study conducted in partnership by the California Coastal Commission and the San Francisco Estuary Institute, utilized historical maps and aerial imagery in order to classify and quantify the extent of habitat change within the Estuary as well as the entire River Valley between 1850 and 2012 (Safran et al. 2017). This study found that between 1850 and 2012, over 42% of the Salt Marsh habitat was lost and 50% of the low marsh-mud flat zone had either been converted to a higher tidal habitat zone lost entirely.

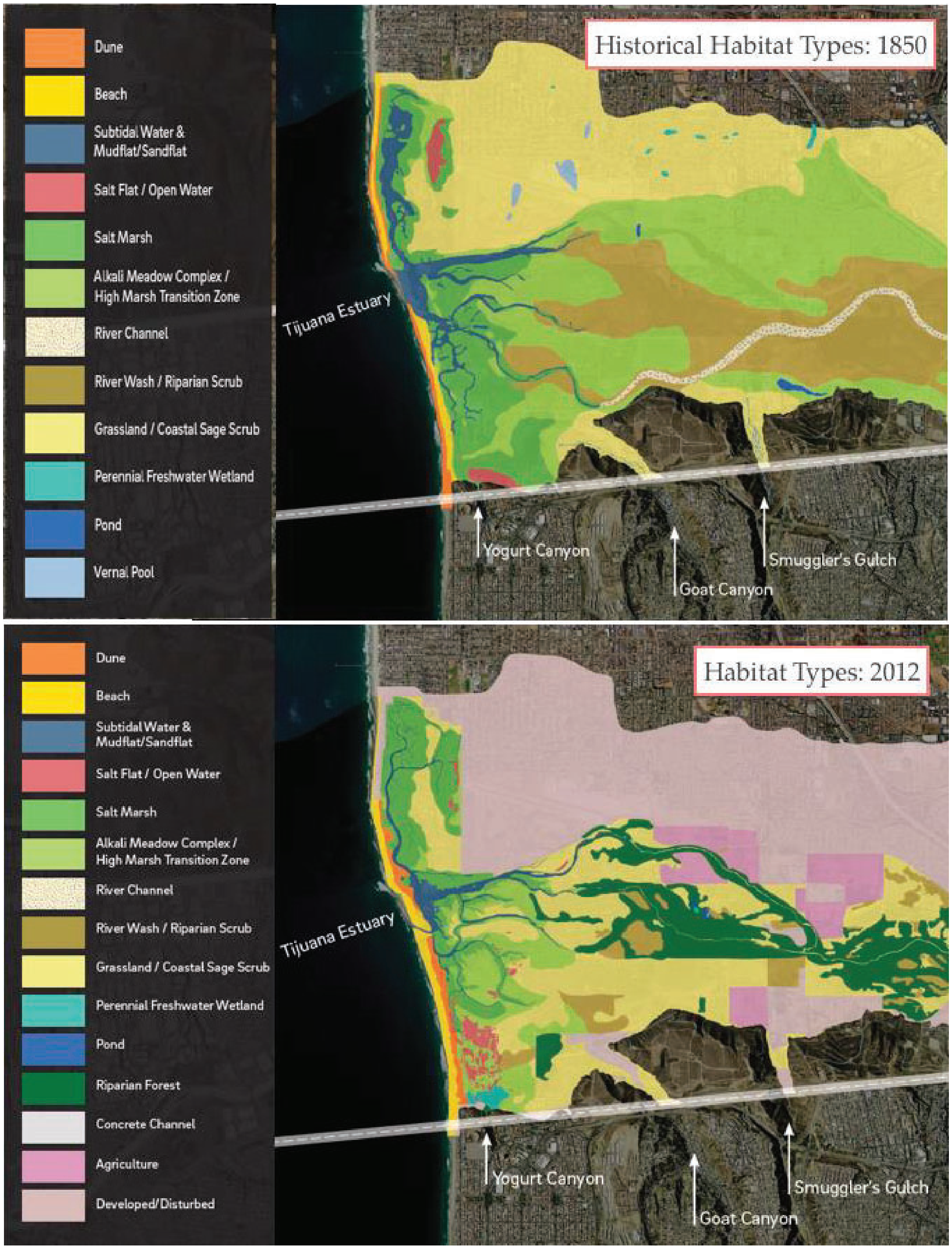

Figure 1.3. Historical Habitat types from 1850 and their contemporary (2012) counterparts. The increase in urbanization in the Tijuana River Valley adjacent to the estuary is illustrated (pink) over the past 75 years. The total area altered from natural habitat to developed land was 1,434 hectares. Figure adapted from Safran et al. 2017.

In estuarine environments, elevational changes can directly influence habitat availability (Zedler et al. 1999). Ward et al. (2003) quantified the sediment accumulation rates in the low marsh platform between 1997 and 1998 to be approximately 12.5 cm/yr. The TRE has two main branches of hydrologic variance, the Northern and Southern arms. Vertical accretion in the northern end of the marsh was quantified by Zedler (1983) between July 1979 and September 1980 as 5 cm. This quantity was calculated by elevation surveys. The study concluded that most of the vertical accretion was associated with winter storm sediment loads. Estuarine habitat zones rely on a narrow margin of elevation in order for plants to survive and thrive. Halophytes, or salt tolerate plants, require a specific soil salinity as to not be out competed by less salt tolerant species (Zedler et al. 2001, Zedler 2010). The transition of marsh platform from low to high marsh is natural in most coastal plain systems, however due to the high rates of sedimentation in Tijuana, this accretion is succeeding over natural rises in sea level and therefore, the rate of habitat type conversion may be happening open a much faster scale than natural.

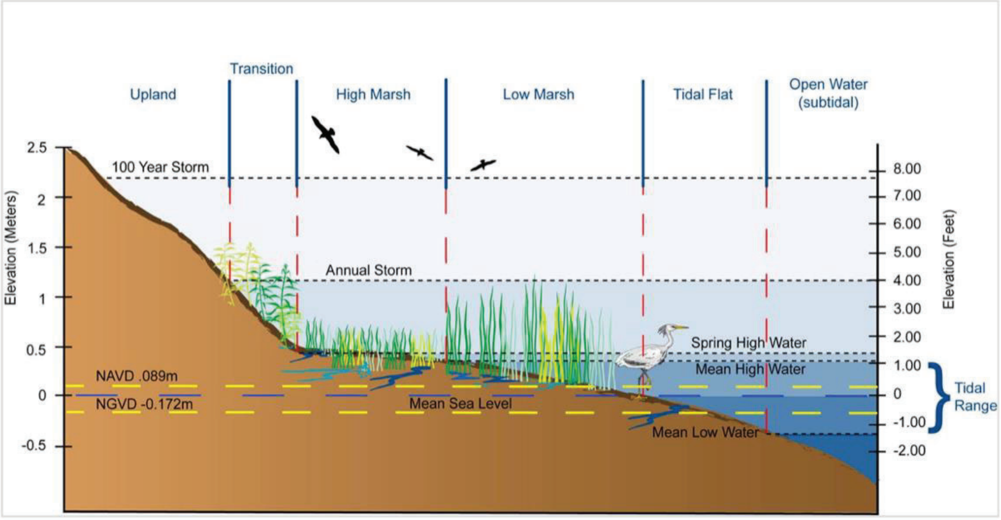

Figure 1.4. Succession of habitat types in coastal estuarine habitats in relation to both the tidal inundation, elevation and annual storm surge heights. (Figure from Titus and Wang 2008).

### *1.3.2. Tidal Prism*

An important hydrologic characteristic of coastal estuaries, tidal prism, can be defined as the volume of water that floods the intertidal platform and recedes back between mean high tide and mean low tide (Lukitina 1998, Vandenbruwaene et al. 2012). Tidal prism refers to the mean volume of water flowing in and out of the estuary in relationship to tides that covers a particular extent of the internal estuarine channels, basins and floodplains (Lukitina 1998, Vandenbruwaene et al. 2012, Zedler 1992). The reach, extent and influence of tidal prism is also directly influenced by elevational changes. The TRE has long been characterized as having a low tidal prism, which has been decreasing over the past few decades (Zedler 1992, Florsheim et al. 1991). It has been highlighted in many studies in the TRE (Zedler 1977, Callaway et al. 2004, Cahoon et al. 1996) and other estuarine systems (Vandenbruwaene et al. 2012, O'Laughlin and van Proosdij 2013) the important influence of tidal prism for assessing overall health and vegetation indices. A hydrologic analysis in 1987 (Williams and Swanson) found an 80% decrease in the tidal prism of the estuary over a 134-year period. Wallace et al. 2005 quantified the tidal prism has been reduced from 18,340 $m^3$ to 14,630 $m^3$ and attributed 88% of this loss to sedimentation along the marsh platform. No uniform quantitative measurements of tidal prism in the TRE have been quantified since this time.

### *1.3.3. Grain Size and Metal Concentrations*

It has been well documented that in the TRE, fluvial sediment deposition is a primary challenge for management (Wallace et al. 2005, Callaway and Zedler 2004, Greer and Stow 2003, Warrick et al. 2012). Much of the sedimentation can be attributed to increases in urban development across the U.S.-Mexican border in the city of Tijuana, Mexico (Weis et al. 2001, Gersberg et al. 2004), urban development can leave large swaths of graded bare Earth exposed and available for transport during episodic rain events. Historically, such a high influx of sediment prompted construction of two catchment basins, designed to trap 30,000 $m^3$ of sediment annually. Sediment is expensive to excavate and remove from these basins, as well as contains a high percentage of fine-grained sediments which are not considered suitable for beach nourishment (Warrick et al. 2012, TRNERR Management Plan 2010). Much of the incoming sedimentation to the TRE are fine-grained particles (Warrick et al. 2012, Wallace et al. 2005). Fine-grained sediments are defined as sediment particles whose diameter is less than 63 μm, and these particles play an important role in nutrient cycling within coastal and estuarine ecosystems (Warrick et al. 2012).

In addition to sediment, the Southern California Coastal Water Research Project (SCCWRP) indicated that a lack of pollution prevention measures has allowed for raw sewage and nonpoint source pollution to cross the international border from the city of Tijuana, Mexico, for decades (SCCWRP 1992, Ganster 1996, Weis et al. 2001), and there is little published data on the retention rates of these pollutants in the estuarine sediments and system. Few studies recent studies document and quantify the pollutant load in the sediment profile of the TRE marsh platform. Weis et al. (2001) collected sediment cores in the northern arm of the estuary and analyzed them both for grain size and heavy metals (specifically Lead (Pb), Copper (Cu), Zinc (Zn), Nickel (Ni), Cadmium (Cd), Chromium (Cr), and Mercury (Hg)). They found metal concentrations to be similar to those found in four previous studies (CSWRCB 1996, Meyer and Gersberg 1997, Gersberg et al. 1989, Sankey 1980), all conducted in the northern arm of the estuary. When compared to other estuaries known for high anthropogenic pollution, they found the concentrations in the TRE to be on the low end of the range. Weis et al also concluded that clay content to be significantly correlated with most metals, except Chromium (2001).

There is little available data on metals and other pollutants in the estuary sediments since the study conducted by Weis et al. in 2001; this is an almost 20-year gap in data collection. Of the studies conducted, none have focused their sampling efforts in the southern hydrologic arm of the estuary, which receives a greater input of fluvial sediment and debris (Warrick et al. 2012). The hydrologic regimes between the northern and southern arms of the estuary are distinct in nature, and the hydrologic inputs are variable. Historical data identify numerous pollutants and a continuous influx of sediment, largely coming from wastewater treatment plants along border with Mexico and the Southern Canyons (Goat and Yogurt) (Biggs et al. 2015). Because of this, gaining a better understanding of the concentrations of pollutants, such as metals, would be useful for effective long-term monitoring, management and restoration in the southern arm of the TRE. Planned restoration projects and ongoing programs such as the Tijuana Estuary Tidal Restoration Project (TETRP) are looking to restore tidal linkages in the system. Restoring tidal linkage is a common restoration tool in Southern California marshes, however, these projects usually involved large scale dredging and sediment excavation (Elwany et al. 2011). Previous TETRP projects in the Northern Arm restored the tidal linkage of the Oneonta Slough (1997) which required sediments to be displaced (TETRP 2011). A stronger understanding of sediment characteristics and metal concentrations will aid in longer term implementation of proposed restoration projects.

### *1.3.4. LiDAR and Imagery Classification*

In the TRE specifically, previous studies sought to understand the extent and ecological make-up of vegetation communities, as well as shifts in these communities from salt to fresh water tolerate species and the influence of invasive (Zedler 1992, O'Brien and Zedler 2006). While these studies provide detailed information about the vegetation make-up, they typically involved extensive field work, such as transects, which can be costly and time intensive (Zheng et al. 2016). In an extensive cooperative research study, The Tijuana River Valley Historical

Ecological Investigation from the San Francisco Estuary Institute (TTRVHEI) (2017), utilized a combination of historical imagery, mapping, and transect derived habitat data to map both the historical and current habitat zones of the Tijuana River Valley. In addition to TTRVHEI, many other studies globally have utilized medium and high-resolution aerial imagery in order to better examine and understand natural processes occurring in coastal and estuarine systems. Several studies have also employed Light Detection and Ranging (LIDAR) acquired data in order to assess the levels that urbanization in a region has had on the fluvial hydrology and channel morphology of the estuarine or riverine system (Notebaert et al. 2008, Nelson et al. 2006, Gilvear et al. 2004, Vandenbruwaene et al. 2013). Vandenbruwaene et al. (2013) utilized both black and white aerial photographs as well as false color infrared photographs in order to extract information about changes to marsh platforms as well as map and extract information about the tidal channel networks. Additionally, they coupled information extracted and classified from aerial photographs with elevation information extracted from digital elevation models (DEM). Using both a DEM from historical topographic surveys as well as a LiDAR derived DEM they were able to extract the aerial extent of low to high marsh transition zones over time; these methods are similar to those employed in our research project here (Vandenbruwaene et al. 2013, Chapter 4, this volume). Los Peñesquitos Lagoon is a nearby southern California coastal estuary located in northern San Diego County where Greer and Stow (2003) also used both analog and digital aerial imagery to map habitat type conversions from 1928 to 1999. The high spatial and vertical resolution of LiDAR imagery offers a very promising information source for scientists and can be utilized to better model temporal change in dynamic fluvial systems. As mentioned, traditionally surveying methods can be costly, time intensive, and often invasive as they require a lot of “on the ground” work within a habitat zone. Having access to high resolution data can allow researchers the opportunity to observe these systems and spend less time in the field (Notebaert et al. 2008). Sequential LiDAR data has been shown to provide good foundational evidence for fluvial and floodplain studies, and Notebaert et al. (2008) found that sequential datasets can be used to quantify sedimentation rates and sediment budgets.

### *1.3.5. Timeline of historical management and restoration efforts*

The TRE is in encapsulated within the Tijuana River National Estuarine Research Reserve (TRNERR). It is one of 29 designated reserves a part of this network that was created as a part of the United States Coastal Zone Management Act of 1972. The reserves were designed to establish long term monitoring, create education centers, and foster environmental stewardship. The TRNERR is a partnership between the United States Federal government and agencies like the National Oceanographic and Atmospheric Association (NOAA) and U.S. Fish and Wildlife and with the State of California agency California State Parks. The “model marsh” or “friendship marsh” is an 8 hectare restoration project conducted in the early 2000’s in the TRE, the primary goal of the project was to restore an area of historical salt marsh habitat that had been converted to a high marsh habitat (Wallace et al. 2005, Safran et al. 2017).

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# 2. Chapter 2

## 2.1. RESEARCH OBJECTIVES AND THESIS STRUCTURE

### *2.1.1. Research Objectives*

The aim of this study is to identify and characterize key geomorphic properties and relationships within the study site. This is a first step in contributing data to and assisting current and future restoration projects in employing a holistic management approach. The study identifies specific physical parameters, grain size and metal concentrations, within the estuary whose relationship to one another is not mutually exclusive and therefore should be addressed in tandem, and tests a remote sensing method for use as a tool for monitoring habitat changes in the estuary.

## 2.2. Thesis Structure

This thesis is presented in two separate chapters that have been prepared as manuscripts for publication. Chapter 3 addresses research questions 1 through 3 that focus on characterizing sediments and quantifying metal concentrations. Chapter 4 will address research questions 4 through 6 which emphasizes the spatial relationship between vegetation and elevation of the marsh platform. Supplementary data, figures and tables have been collated and will appear in the appendices of this thesis.

## 2.3. Research Questions

1. What are characteristics of the sediment, specifically the grain size distribution and metal concentration, in the southern arm of TRE?
2. How are median grain sizes and metal concentrations of the sediments within the estuary related?

3. How do the metal concentrations compare to environmental screening values and geogenic data?

4. Can unsupervised classification of elevation data be used to classify and map vegetation habitat types within the estuary?

5. How have the habitat types changed over time?

6. Where has the Tijuana River Estuary experienced erosion and deposition over the same time period?

# 3. **Chapter 3:** Understanding Grain Size and Metal Distributions in the Tijuana River Estuary

*This chapter has been formatted for publication in the Journal of Coastal Research.*

## 3.1. Abstract

Coastal wetlands and salt marshes are among the ecosystems most impacted by anthropogenic activity. Estuarine sediments serve as an important sink and source for both nutrients and metals. In excess, some of these metals can be hazardous to both environmental and human health. We quantified the relationship between the distribution of sediment grain size and the metals in the Tijuana River Estuary (TRE) in San Diego County. For this study, 78 sediment samples from the tidal channels and low marsh portions of the TRE were collected for grain size and metals analysis using a Particle Size Analyzer (PSA) and an X-Ray Florescence (XRF) unit, respectively. The results of the metals analyses were compared to the Screening Quick Reference Tables (SQuiRTs) developed by NOAA, the California Regional Water Quality Control Board, San Diego Region Order No. R9-2014-0041 Conditional Waiver Number 10, and average background and geogenic data on metal concentrations within the region (Buchman 2008). Sediment grain size was significantly correlated with metal concentrations of all metals tested, and Copper had the strongest relationship ($R^2$=0.77, P-Value <2.2e-16). Sample concentrations for Arsenic, Chromium, and Lead were all higher than the CRWCB Tier 1 Inert Soil Waiver 10 screening levels concentrations. Geogenic concentrations of all five metals tested were found to be lower than the 95% confidence value for all our samples. Widespread sedimentation, largely from terrigenous input, has been documented throughout the study site over the last half-century. Our research suggests that further analysis and long-term monitoring to better understand the potential risk and mobility of metal concentrations in the sediments is recommended for ecosystem management.

**ADDITIONAL INDEX WORDS:** Sediments, Metals, XRF, Grain Size, Estuary

### 3.2. Introduction

Coastal wetlands and estuarine ecosystems are unique and vital ecosystems; they exist at a critical and often diverse transition zone between the land and sea. Their unique location lends itself to high rates of both economic and environmental productivity (Barbier et al., 2011; Li et al., 2018). Coastal ecosystems are among the most anthropogenically impacted; and 50% of global salt marshes have been either lost or severely degraded as a result of human activities (Zedler, Norby, and Kus, 1992). On the west coast of the United States, 90% of wetlands and coastal ecosystems have been completely destroyed or modified from their natural state (Barbier et al., 2011; Li et al., 2018).

Rapid industrialization, urbanization, and global increases in agricultural practices have increased the pathway for many pollutants to enter the environment. Due to their proximity to high population urban and economic centers, coastal ecosystems are particularly vulnerable to pollutants such as metals and hydrocarbons (Gargouri et al., 2018; Sakan et al., 2014; Venkatramanan et al., 2018). The chemical and physical properties of coastal estuarine sediments may allow them to serve as a sink or source of metal contaminants due to their adsorption, precipitation, and chemical properties (Gargouri et al., 2018). The variability of metals and their ability to bond, mobilize, and persist in the environment has been shown to be closely related to the grain size, organic carbon content, minerology, and hydrodynamic conditions of the sediment substrate (Gargouri et al., 2018). The organic carbon content has a strong relationship with sediment grain size and is often correlated with metal concentrations. Lin et al. (2001) tested the relationship between grain size, organic carbon and heavy metals from pollution sourcing in the Yangtze River, China, where they found that while both grain size and organic carbon content were significantly correlated with higher metal concentrations, grain size was the stronger determining factor in this relationship for all metals tested in their study (Iron, Manganese, Zinc, Copper, Lead, Cadmium).

Metals are among the most persistent pollutants in sediments, water, and even ecosystem biota (Gargouri et al., 2018; Liang et al., 2018). The tendency of metals to persist and accumulate in the sediments of marine systems can cause them to reach potentially toxic or hazardous levels for the benthic and sediment dwelling organisms, which can have impacts up the ecosystem food chain (Jones and Turner, 2010; Sakan et al., 2014). Copper, Zinc, and Lead are commonly found as components of anti-fouling paints, used in many marinas, and on boats and hard-standing infrastructure (Jones and Turner, 2010). Due to their wide-spread use and proximity to the marine environment, their interactions in these environments have been extensively studied. Bivalves and burrowing organisms have served as reliable bio-monitors for benthic marine ecosystems; and are often used in studies to evaluate estuarine communities (Boening, 1999; Fukunaga et al., 2010). These organisms are good monitors because they are in contact with the surface sediments, have low mobility, generally have long life cycles, and are known to accumulate metals (Boening, 1999). For example, persistent toxic metals and pollutants such as Copper, Lead, and Zinc can persist in the tissues of marine organisms and have negative impacts on species richness and community structure (Fukunaga et al., 2010). Understanding metal concentrations in these systems is particularly valuable due to various biogeochemical mechanisms that can make metals highly mobile and subject to both biomagnification and bioaccumulation in the environment (Raj and Jayaprakash, 2008). High concentrations of metals can be potentially toxic and hazardous to both human and ecosystem health. Thus, it is critical for restoration and ecosystem land managers to know the potential for mobility of the metals and the sediments (Venkatramanan et al., 2018) and to do so, they must first quantify the grain size distribution and metal concentrations present in the system of interest. Understanding not only the presence of potential pollutants, but also their location and extent is important for effective site management; this study seeks to identify some of these key components for the area of study within the Tijuana River Estuary.

### *3.2.1. Study Site*

The Tijuana River Estuary (TRE) is located in southern San Diego County, California. TRE encompasses 1000 hectares of reserve ecosystems and southernmost extent is adjacent to the United States-Mexico International border (Figure 3.1). The TRE lies at the mouth of the 1,750 $mi^2$ Tijuana River Watershed, and three-fourths of this watershed lies within Mexico's border. The TRE is classified as a tidal coastal plain estuary at its western extent where its inlet meets the Pacific Ocean, and as it moves east, the ecosystem grades into a river valley floodplain estuary (Zedler, Norby, and Kus, 1992). The TRE is categorized as an intermittently open estuary, although it closes seldomly. Most recently the mouth closed in 2016, this was speculated to be related to beach nourishment projects to the north, sediment was carried down due to coastal longshore drift currents, this was the first time the mouth closed since 1983 (Sarah Giddings Lab, 2016). Hydrologically, the TRE is influenced by diurnal coastal tidal inundation and input as well as and freshwater inputs which are from both the Tijuana River and rainfall events within the watershed. Natural ecosystem functions of estuaries are largely influenced by physical parameters, such as tidal input and sediment transport. In Southern California, restoration projects in estuarine systems typically involve reestablishing tidal flow (Elwany et al., 2011). The Tijuana Estuary Tidal Restoration Program is an example of a long-term restoration program in the study site that has worked to restore this tidal linkage in different parts of the system (TETRP 2011). The TRE has been subject to degradation and pollution due to a high influx of sediments and raw sewage entering the estuary from its adjacent ephemeral tributaries along the Mexican Border, as well as from the City of Tijuana (Ganster, 1996; Weis, Callaway, and Gersberg, 2001). The estuary is split into two distinct hydrologic arms, the Northern Arm which includes the Tijuana Slough National Wildlife Refuge, and the Southern Arm which includes the Model Marsh, Sediment Basins, Goat and Yogurt Canyons. (Figure 3.1b). The northern arm receives its main freshwater inputs from the Tijuana River, the Southern arm largely only receives freshwater inputs from ephemeral streams and rain events. The

Northern Arm is also tidally connected to the Oneonta Slough, which was tidally restored in the late 1990's (Safran et al., 2017). This study focused on the southern arm of the estuary, due the lack of recent data on metal concentrations or pollutants and its selection as a restoration site (TETRP, 2011).

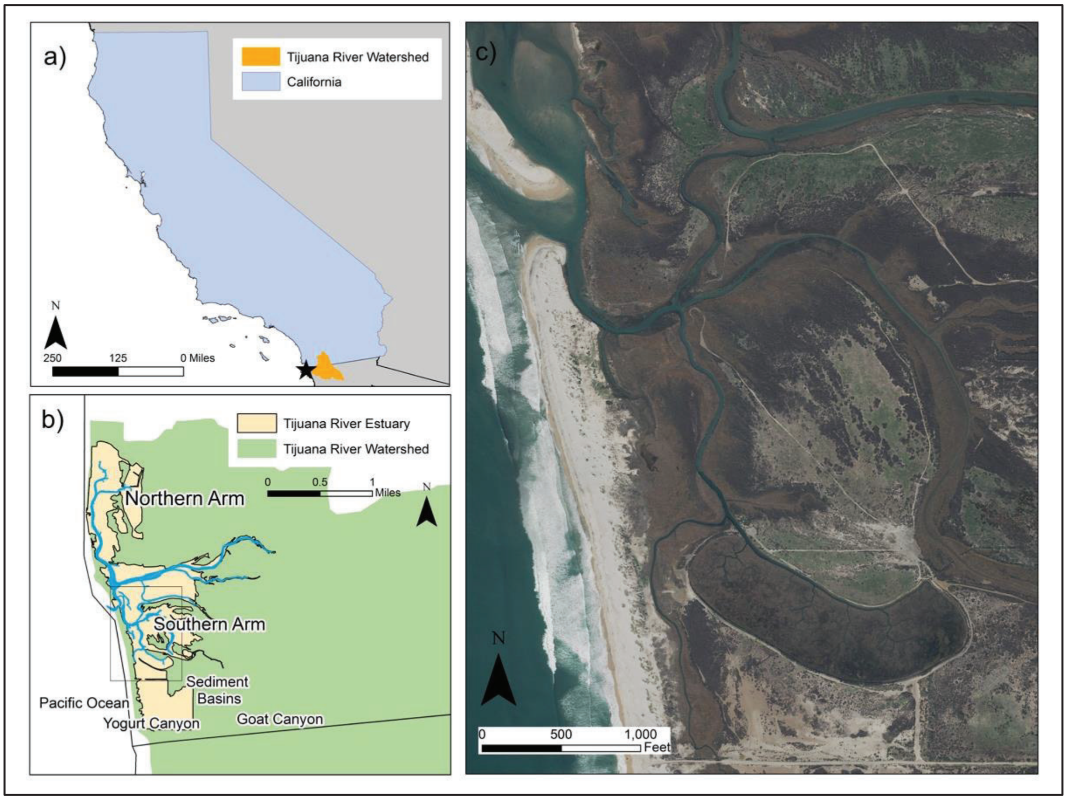

Figure 3.1. a) The Tijuana River Estuary is located in southern California at the mouth of the Tijuana River Watershed, three-fourths of which lies in Mexico. b) The Tijuana River Estuary is comprised of two distinct hydrologic units, the Northern and Southern Arms. c) This research focuses on the Main Channel and the Southern Arm.

While tidal sources of sediments play a role, fluvial sediment deposition from terrigenous sources is the primary influence of sedimentation into the system (Callaway and Zedler, 2004; Greer and Stow, 2003; Wallace, Callaway and Zedler, 2005; Warrick et al., 2012). The San Diego County region has the highest variability in rainfall of any region in the United States and on average receives 25-

30cm of rainfall (Kalansky et al., 2018). A lot of the seasonal variability (80%) is caused by extreme precipitation events, these events are ones that exceed the 95th percentile for volume of precipitation (Kalansky et al., 2018). These large storm events in the region have the capacity to mobilize massive volumes of sediment and debris. In the past, these rainfall events have been coupled with large episodes of sedimentation in the TRE. Cahoon et al. (2004) calculated that the winter storms of 1993 mobilized approximately 5 million metric tons of sediment, a large portion of which (31,941 metric tons) was trapped by the low salt marsh habitats. A large portion of the sediment that is available for transport from these rain events can be attributed to the increased urbanization of the Tijuana River Watershed (Gersberg, 2004; Weis, Callaway, and Gersberg, 2001). Tijuana was the fastest growing city in Mexico between 1990 and 1995, and the population continues to grow (Biggs, Anderson and Pombo, 2015). This increase in population has led to wide-spread development and urbanization. Increases in sedimentation, caused by land use changes induced by urbanization, correspond with Wolman's Cycle of Erosion model (Wolman, 1967). During the landscape construction phase, the model displays exposed surfaces and the production of sediment orders of magnitude greater than natural undisturbed surfaces. The continued growth of Tijuana's population has encouraged continued construction and further urbanization; this results in a consistent source of sediment available for runoff during storm events.

Large events of sediment deposition and accumulation on the marsh platform is a primary management challenge for TRE ecosystem managers. To combat the high influx of sediment and debris, two debris catchment basins were constructed between 2003-2005, with the capacity to trap 30,000 $m^3$ of sediment annually (Warrick et al., 2012). This sediment however is expensive to remove, and also contains a high percentage of fine-grained sediments (very fine sands, silts, and clays) which are not suitable for beach nourishment projects (Warrick et al., 2012). Much of the incoming sedimentation to the TRE are fine-grained particles (Wallace, Callaway and Zedler, 2005; Warrick et al., 2012), defined here as sediment particles whose diameter is less than 63 μm (very fine sand, silts, and clays; Wentworth, 1922). While sediments play an important role in nutrient cycling within coastal and estuarine ecosystems, in excess, they raise the elevation of the estuary platform. Their particle size and the presence of contaminants prevent them from being easily or inexpensively removed and are therefore trucked away or left in large piles near the basins (Warrick et al., 2012). This still poses an issue for land use managers as well as continues to provide a source of terrigenous sediments to the TRE marsh platform.

In addition to sediment, the Southern California Coastal Water Research Project (SCCWRP) indicated that a lack of pollution prevention measures has allowed for raw sewage and nonpoint source pollution to cross the international border from the city of Tijuana, Mexico, for decades (Ganster, 1996; SCCWRP, 1992; Weis, Callaway, and Gersberg, 2001) and there is little published data on the retention rates of these pollutants in the sediments of the TRE and their impact on the overall system. Several studies conducted over the last three decades document and quantify the pollutant load in the sediment profile of the TRE marsh platform. In sediment cores from the northern arm of the estuary, Weis, Callaway, and

Gersberg (2001) found metal concentrations (specifically Lead, Copper, Zinc, Nickel, Cadmium, Chromium, and Mercury) to be similar to those found in four previous studies (CSWRCB, 1996; Gersberg, Trindade, and Nordby, 1989; Meyer and Gersberg, 1997; Sankey, 1980), all of which were conducted in the northern arm of the estuary. This project focused on the regions of the estuary commonly designated as the Main Channel and Southern Arm, in order to expand available data on metal concentrations. When compared to other estuaries known for high anthropogenic pollution, Weis, Callaway, and Gersberg (2001) found concentrations in the TRE to be less than many other estuaries significantly impacted by pollution. This study also concluded that clay content was significantly correlated with most metals, except Chromium (2001).

There is a lack of recent data on metals specific to the Tijuana River Estuary sediments since the Weis, Callaway, and Gersberg study in 2001; an almost 20-year gap in data collection. The Southern California Bight 2008 Regional Monitoring Program, a project conducted by the Southern California Coastal Water Research Project (SCCWRP), collected detailed sediment chemistry for both organics and inorganics throughout the Southern California Bite Region. The study utilized 383 grab samples, some of which were located in the main channel of the TRE (Schiff et al., 2011). While they did not separately analyze the TRE sediments for their study, they did conclude that over the last five-year monitoring term (2003-2008) the percentage of sediment classified in "acceptable sediment condition" decreased in southern California estuaries from 86% to 62% (Schiff et al., 2011). Metal concentrations from the BIGHT program for individual grab samples collected in the main channel of the TRE are available through the California Environmental Data Exchange Network (CEDEN) online database (CEDEN dataset, 2019). The Shiff et al. (2011) study analyzed metal concentrations in sediment using inductively coupled plasma mass spectrometry (ICP-MS), whereas our study measured bulk metal concentrations using X-Ray Fluorescence (XRF). Of the site-specific studies conducted, none sampled in the southern hydrologic arm of the estuary, which receives a greater input of fluvial sediment and debris. The hydrologic regimes between the northern and southern arms of the estuary are distinct in nature, and therefore, the fluvial inputs are variable. Historical data identifies the primary sources of pollutants and continuous influx of sediments as historical wastewater treatment plants along border with Mexico and the southern Canyons (Goat Canyon and Yogurt Canyon) (Biggs, Anderson and Pombo, 2015). Because of this, it would be useful for effective long-term monitoring and management to gain a better understanding of the sediment characteristics and the concentrations of metals in the southern arm of the TRE.

The objectives of this study are threefold: (1) to identify baseline characteristics, specifically, the grain size and metals concentrations, of sediment in the TRE, (2) determine the relationship between grain size and metal concentration in its spatial context, and (3) compare the metal concentrations found in the sediment samples to commonly used screening levels established by the US EPA and NOAA, as well as regional background and geogenic concentrations of metals. These data may be used to identify a need for long-term monitoring, improved management, and restoration planning.

## 3.3. Methods

### *3.3.1. Grain Size and Metal Analysis*

Sampling sites were selected in order to capture both in-channel and floodplain sediments for each cross-section. We collected sediment samples using a hand core, taken from the surface to approximately 6-inches in depth, and stored in Ziploc bags. Sampling sites were selected along each of the cross-sections mapped for accessibility and distribution throughout the main channel and the southern arm of the TRE (Figure 3.2). In total, 78 sediment samples were collected from 12 cross-sections in July 2017 and January 2018. Samples were individually labeled and brought back to the lab for analysis. All sediments were dried in an oven at 105° Celsius, homogenized, and if necessary, disassociated with a mortar and pestle following drying using the methods of Weis, Callaway, and Gersberg (2001).

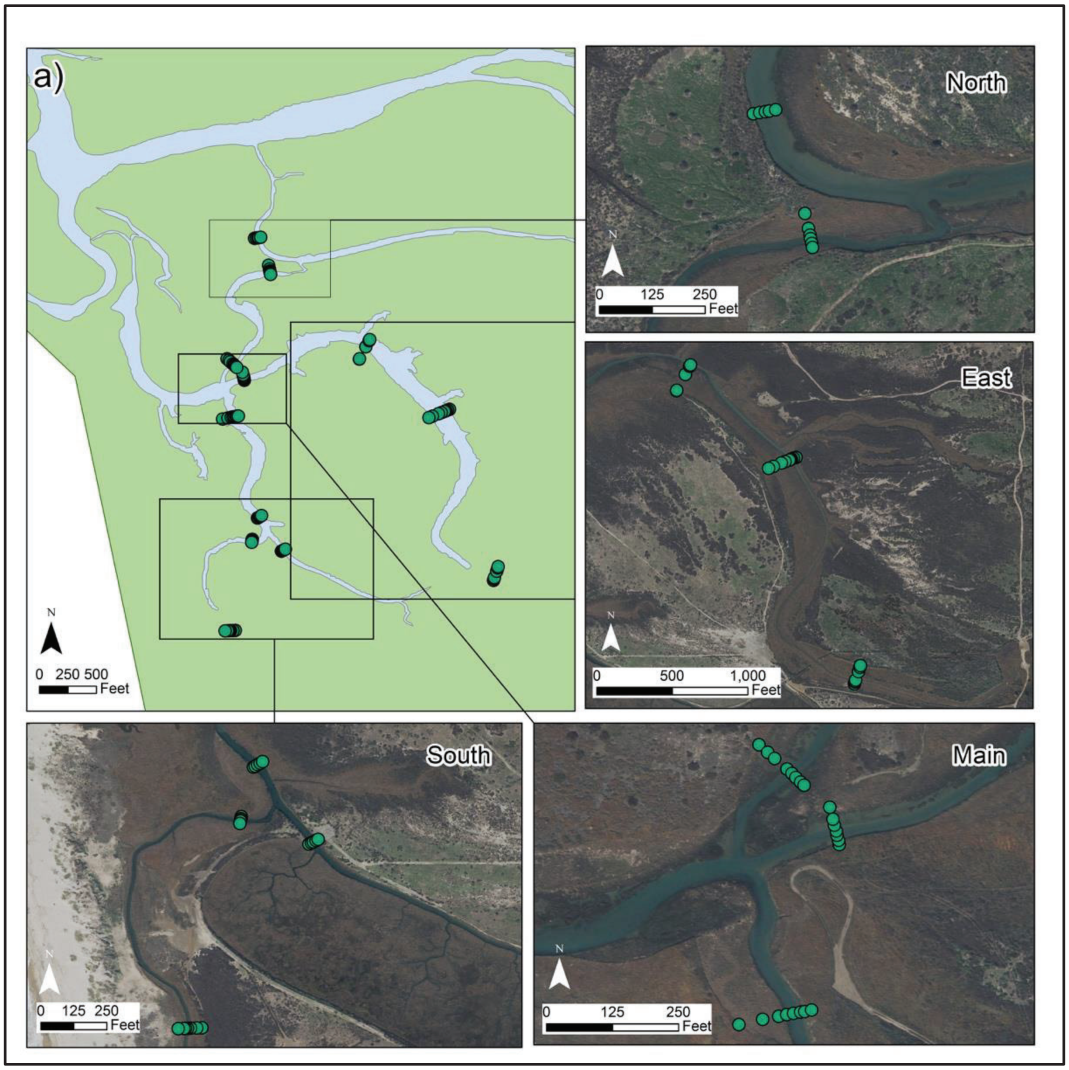

Figure 3.2 a) The Tijuana River Estuary has two distinct hydrologic units, the Northern and the Southern Arm. This study focuses sampling only on the Main Channel and the Southern Arm. Sample sites were grouped into four distinct regions throughout the Main Channel and the Southern Arm shown clockwise from the top: North, East Main, and South. Cross-section sampling locations (green dots) are indicated for each of the four regions.

For grain size analysis, all 78 samples were run on a Cilas 1190 Particle Size Analyzer in triplicates to account for potential heterogeneity within the samples. For each sediment sample, the median grain size values (50th percentile or D50) were calculated. Next, sediment classes were assigned based on the Wentworth Grain Size classification scale (1922).

Sediments were analyzed for metal content, including Arsenic (As), Chromium (Cr), Copper (Cu), Lead (Pb) and Zinc (Zn), using an Innovex X-5000 X-Ray Florescence (XRF) unit, also in triplicate. These metals were chosen for comparison with past studies and are the most appropriate for total metal concentration using XRF.  Following calibration using metal standards. Metals content is expressed throughout this study in parts per million (ppm).

### *3.3.2. Data Processing and Statistical Analysis*

Samples were divided into the four distinct regions within the study area (Fig. 3.2), to utilize sufficient data points for statistical analysis as well to map the spatial distribution of the metal concentrations. The regions were defined by the location of the cross-sections within the tidal channel network. ProUCL 5.1, a comprehensive statistical software package developed by the US EPA for analyses of normal and non-normal distributed environmental datasets, was used for all statistical calculations. A Students-t Test was used to calculate the 95% Upper Confidence Limit (UCL) for the concentrations of each of the five metals (As, Cr, Cu, Pb, Zn) analyzed for each region and for each of the individual cross-sections. Statistical outlier tests were also performed to ensure the values used were an accurate representation of each specific region. R-Studio was used to calculate statistical significance and $R^2$ values of the relationship between grain size and metal concentrations. Metal concentrations were compared the Screening Quick Reference Tables (SQuiRTs) published by the National Oceanic and Atmospheric Administration (NOAA) (Buchman, 2008). These tables provide screening levels based on aquatic species toxicology and are a basic standard of reference for both organic and inorganic materials within marine ecosystems. We compared our findings to the California Regional Water Quality Control Board, San Diego Region Order No. R9-2014-0041 Conditional Waiver Number 10, specifically the collection of dredge material in soil stockpiles and its reuse within San Diego County. Finally, the metal concentrations from each of the four sampling regions were also compared to the mean background metals concentrations throughout southern California (Bradford et al., 1996) and local geogenic concentrations of metals within the Bay Point Formation, which commonly outcrops in areas surrounding the TRE (Harris et al., 2013).

### 3.4. Results

The D50 values for all samples ranged between 7.96 µm and 670.15 µm, with a median of 24.90µm. Of the 78 samples, 66, or 85%, had D50 values that were less than 63 µm and classified as very fine-grained. Each of the five metals (As, Cr, Cu, Pb, Zn) showed a significant inverse correlation with sediment grain size. Higher concentrations for all metals were strongly inversely correlated with finer grained samples (Figure 3.3). Cu concentrations exhibited the strongest correlation with grain size ($R^2 = 0.7717$, p-value < 2.2e-16), followed by Zn, Cr, Pb, and As respectively (Table 3.1). Of the 12 coarse-grained samples, none of the five metals exceeded the Effects Range Low (ERL) for marine sediments (SQuiRTs) or Conditional Waiver 10 for Tier 1 inert soils. When assessed spatially, the samples containing highest metal concentrations were confined to the samples collected on the floodplain. This was consistent for all metals Chromium, Copper, Lead and Zinc and excluded arsenic.

Table 3.1. $R^2$ values of metal concentrations versus grain size for all samples for each metal sampled. P-values were obtained using a power regression model.

| METAL | $R^2$ OF METAL CONCENTRATION VS. GRAIN SIZE (LOG) | P-VALUE |
|---|---|---|
| **ARSENIC (AS)** | 0.5442 | 1.965e-14 |
| **CHROMIUM (CR)** | 0.6528 | 5.902e-15 |
| **COPPER (CU)** | 0.7717 | < 2.2e-16 |
| **LEAD (PB)** | 0.5585 | <2.2e-16 |
| **ZINC (ZN)** | 0.7657 | <2.2e-16 |

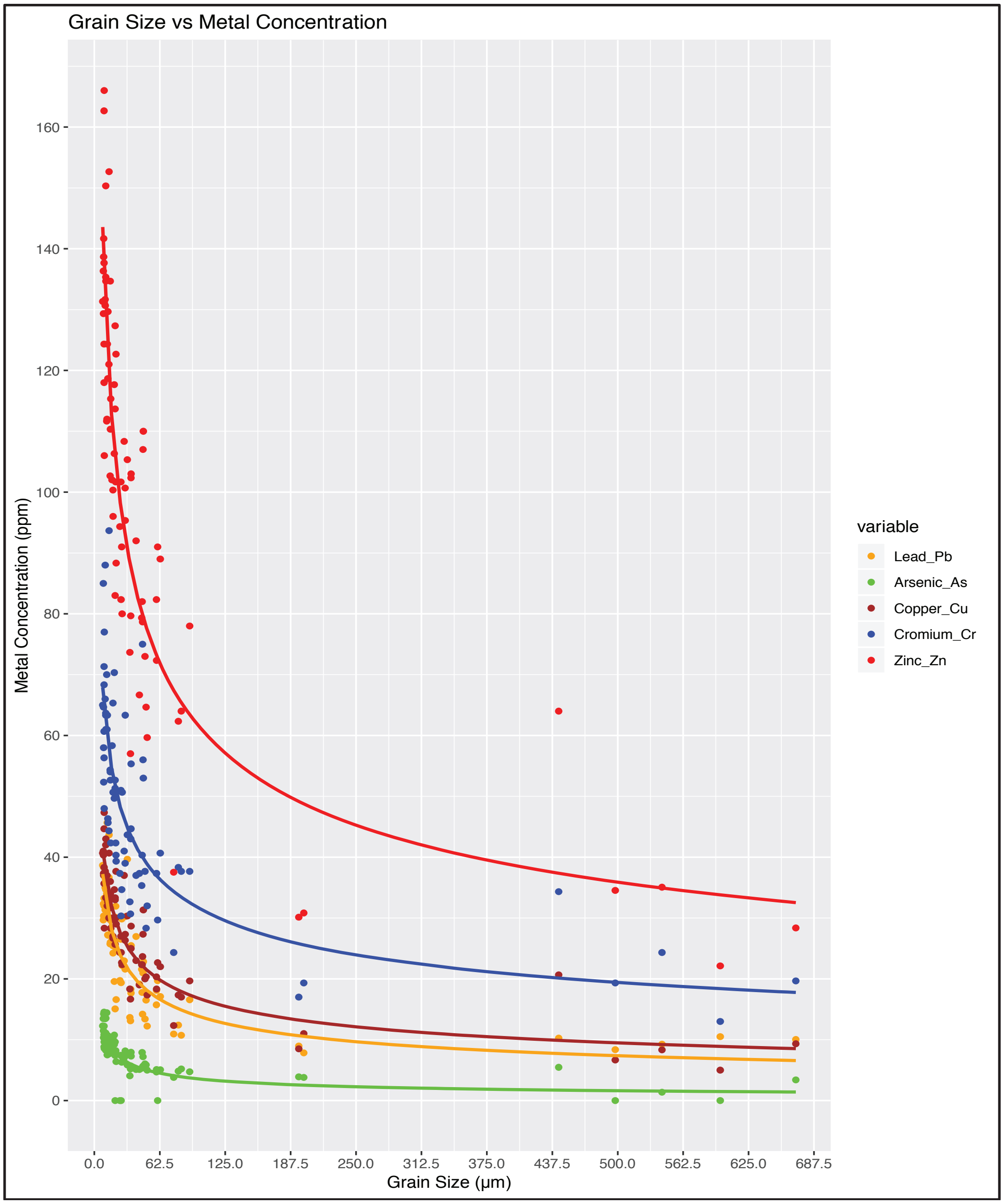

Figure 3.3. Relationship between grain-size (in µm) (x-axis) and concentration (in ppm) of each of the five metals tested (As, Cr, Cu, Pb, Zn) (y-axis).

The samples were further analyzed both by region and by cross-section, in order to account for potential outliers as well as address any spatial disparities. The 95% confidence intervals of the metal concentrations were calculated using a Students-T Test (Table 3.2). Concentrations of As for all regions were higher than both the CRWCB Tier 1 and ERL SQuiRTs minimum values. Concentrations of both Cr and Pb exceeded CRWCB Tier 1 screening levels but did not exceed the ERL. The maximum concentrations of all metals were found in the East region. Concentrations Cu and Zn did not exceed either CRWCB Tier 1 or ERL levels at any sampling region.

Table 3.2. Tier 1 and Effects Range Low (ERL) values compared against metal concentrations for each of the four sampling sub-regions.

| METAL | TIER 1 | ERL | EAST | MAIN | SOUTH | NORTH |
|---|---|---|---|---|---|---|
| **ARSENIC** | 3.5 | 8.20 | 10.61 | 8.48 | 8.30 | 8.27 |
| **COPPER** | 60 | 34 | 33.70 | 32.81 | 27.79 | 31.05 |
| **CHROMIUM** | 50 | 81 | 59.72 | 53.36 | 50.75 | 55.27 |
| **LEAD** | 15 | 46.70 | 32.42 | 27.40 | 22.01 | 30.30 |
| **ZINC** | 149 | 150 | 119.20 | 116.5 | 95.48 | 114.90 |

Geogenic data comparisons were conducted for all samples, these values were used in conjunction with Geogenic data (Harris et al., 2013) (listed in Table 3.3). The geogenic data set is a 95% UCL concentration calculated from sediment and rock samples from the Bay Point Formation, a sedimentary rock formation found in the San Diego Embayment Region which commonly outcrops in the southern canyons of the TRE associated with the sedimentation runoff (Warrick et al., 2012; Harris et al., 2013). The concentrations for all five metals tested in this study are higher than the geogenic metals average values found in the Embayment Region. Our samples were also compared to the mean background concentrations published by Bradford et al. (1996) (Table 3.3, column 2). When compared to the background concentrations, only Arsenic registered as having a higher concentration value than the background concentration value.

Table 3.3. Geogenic and background concentrations of metals compared to 95% confidence values for all sediment samples (this study). [1]Harris et al. 2013; [2]Bradford et al. 1996.

| METAL | 95% UCL Values of Geogenic Metal Concentrations (PPM)[1] | Mean Background Metal Concentrations[2] | 95% UCL (PPM) All Samples |
|---|---|---|---|
| **ARSENIC** | 2.50 | 3.50 | 8.04 |
| **COPPER** | 7.10 | 28.70 | 29.50 |
| **CHROMIUM** | 16.69 | 122.00 | 51.32 |
| **LEAD** | 3.60 | 23.90 | 26.01 |
| **ZINC** | 39.30 | 149.00 | 104.60 |

## 3.5. Discussion

Of the samples we collected, the majority of them had a D50 or median value that was classified as fine-grained, with only 12 of the 78 samples considered coarse grained, according to the Wentworth scale (1922). Spatially, the sediment samples we found to have the smallest median grain sizes were most commonly located in the flood plain. However, the samples were largely homogeneous in their spatial distribution. Of the samples not considered fine-grained, all but two were confined to the channel. This is consistent with many other fluvial systems, where location of sediment grain size is largely correlated with flow rates and patterns, longitudinally and laterally, of that fluvial system. Fresh-water inputs to the southern arm are largely ephemeral in nature, dominated by seasonal rain events, and, thus, hard to quantify annually. Deposition and mobilization of sediment is controlled by the range of energy characteristics in a system; the more energy, the larger particle grain size a system can transport (Watson et al., 2013). The TRE receives both tidal and freshwater inputs, however, this system has long been classified as having a low tidal prism, and has since been decreasing (Zedler, Norby, and Kus, 1992; Florsheim et al., 1991). Tidal prism can be defined as the volume of water that floods the intertidal platform and recedes back between mean high tide and mean low tide (Lukitina, 1998; Vandenbruwaene et al., 2012). In the TRE, the tidal input of sediment is low, and a reduction in the tidal prism may also be coupled with a reduction in marine and coast sourced sediments, which tend to have a larger grain size (beach sands) (Warrick et al., 2012). This may account for why, even closer to the tidal inlet, we did not obtain many samples with larger median grain sizes. Most of the sedimentary input into the system is from terrigenous sources, and much of this sediment deposition occurs during large storm events (Warrick et al., 2012). This deposition is occurring across the marsh platform (see chapter 4, this volume), and the homogeneity in the deposition may account for the lack of spatial variability in sediment grain size throughout the study site. This finding could be utilized in preparing for restoration projects related to grain size, sediment transport, and tidal prism, such as TETRP (2011).

Our study revealed that fine-grained sediments were inversely correlated with all five of the metals we tested (Arsenic, Chromium, Copper, Lead, and Zinc). The relationship between grain size and metal concentration was found to be highly significant for all five metals. Similarly, Lin et al. (2001) found grain size to be the strongest regulator when assessing the spatial variability of metals entering the marine and coastal environment. Both metal concentration and organic carbon content were positively related to grain size, but they determined that this was still subsequently controlled by grain size. These higher metal concentrations were also directly correlated spatially with terrigenous sediment runoff into the system (Lin et al., 2001). While our data suggests a terrigenous source, further analysis of sediments at input sites into the estuary is needed to identify specific sources of the sediments in the TRE.

Metal concentrations in the coastal marine environment have been studied globally, and this has led to the development of many pollution indices and screening tools for various environments (Gargouri et al., 2018). These indices are a useful tool for both ecosystem and human health risk managers. We compared our concentrations to the NOAA SQuiRTs Tables and California Regional Water Quality Control Board (CWRCB) Tier 1 indicators. The SquiRTs tables were specifically designed only as screening tools for marine and freshwater sediments, and they indicate the potential for hazardous pollutant quantities. This is important to our site, as increased levels of coastal urbanization have increased the accessibility for pollutants to enter the marine environment (Jones and Turner, 2010; Raj and Jayaprakash, 2007). Persistent toxic metals and pollutants such as Copper, Lead, and Zinc can remain in the tissues and shells of sediment dwelling marine organisms, such as bivalves, and this can have negative impacts to both the species richness and community structure (Fukunaga et al., 2010)

We found that our values for Arsenic, Chromium, and Lead were all higher than the CRWCB Tier 1 Inert Soil Waiver 10 screening levels concentrations. The Tier 1 waiver is specifically for waste pile disposal within San Diego County and may only apply if the sediments from the area are dredged and relocated outside of the estuary. Restoring tidal linkage is a common restoration technique, particularly for Southern California estuaries (Elwany et al., 2011). However, while it may improve tidal linkage, there are also high costs and challenges associated with dredging projects (Elwany et al., 2011). We chose to use this screening tool due to the nature of past and future potential restoration projects in the TRE. The Tijuana Estuary Tidal Restoration Program (TETRP) is a long-term, multi-phased program that is working towards restoring more than 500 acres of wetland habitat with the primary motivator of restoring important tidal linkages in the TRE system. Past TETRP restoration projects that have involved large scale excavation and dredging of sediments such as the Oneonta Tidal Linkage (1997) Model Marsh Restoration (1999-2000) (TETRP, 2011). Based on the results of this study, future restoration projects focused on flow modeling would benefit from knowing both the grain size data and homogeneity of the sediments, while those that are considering dredging may need to include more detailed analysis and processing methods for metal concentrations to determine where the dredged material can be deposited.

The concentration of all metals tested in our study site were higher than the average geogenic concentrations of metals calculated from the nearby sedimentary rock formations. The Bay Point Formation is a sedimentary rock formation that commonly outcrops in the surrounding areas to the TRE. Many exposed cliff faces can be seen in the upper part of the Tijuana River watershed, as well as those that outcrop on the two southern canyons, Goat Canyon and Yogurt Canyon, play a role in sediment transport to the TRE. These canyons come into the United States across the border from Mexico and enter the Tijuana River Estuary, providing a source for large volumes of incoming sediment and debris into the TRE, typically during large storm events (Biggs, Anderson, and Pombo, 2015; Biggs et al., 2010; Kennedy and

Tan, 2008; Taniguchi and Biggs, 2015; Wallace, Callaway and Zedler, 2005). The relatively high concentrations of metals found in our samples throughout the study area could be related to sedimentary and geomorphic processes; whereby sediments eroded from hillslopes and other areas through steep canyons are rapidly deposited in low slope, slow water areas into which they flow, such as estuaries, as in this case. These processes, repeated over time, could result in a higher than normal accumulation of the metals. In addition to their geogenic characteristics, these canyons are more intimately connected to the larger Tijuana River Watershed system. The increased sediment runoff from urbanization in Tijuana, Mexico, has large down-stream impacts on the natural ecosystems (Biggs et al., 2010) which is our study site, the TRE (Callaway and Zedler, 2004).

Our study indicates that there is an increased need for long term sediment monitoring throughout the study site. There has been an almost 20-year gap in data collection in the estuary, with the last focused study in 2001 (Weis, Callaway, and Gerberg). The Southern California Bight 2008 Regional Monitoring Program collected a few grab samples in the main channel of the TRE as part of a larger southern California sediment study. Findings of metal concentrations from sediment samples collected throughout the TRE in the past nearly 40 years are shown in Table 3.4. There is variability in both the sampling methodology used as well as the analytical methods to assess trace metal content. For example, some sampling methods included taking deeper cores versus surface grab samples, and this coupled with analytical variation in methodology may account for the variability observed. The results in the table show years with higher metal content and years of lower metal content. This may indicate that the season in which the samples were collected play a role in concentrations of metals obtained or that climatic variability in southern California has an influence on the deposition and longevity of metals remaining in the sediment. This variability in results and lack of sampling frequency provide a strong argument in favor of long term, seasonal monitoring throughout the study site. While comparable, individual studies used variable methods for measuring metal content, and it is notable that the Schiff et al. (2011) results from 2008 were substantially lower than any other study, possibly due to the analytical methods used in this study. Our study is the only one that collected samples in the southern arm of the estuary, all others with the exception of Schiff et al. (2011) which included the main arm, were located in solely the northern arm.

Table 3.4. Metal concentrations of surface sediments in the Tijuana River Estuary in Parts Per Million (PPM), na indicates unavailable data. [1]Sankey, 1980; [2]Gersberg, Trindade, and Nordby, 1989; [3]CSWRCB, 1996; [4]Meyer and Gersberg, 1997; [5]Weis, Callaway, and Gersberg, 2001; [6]Schiff et al., 2011; [7]this study. [*]ICP-MS, [+]XRF, [#]Atomic Absorption Spectrometry (AAS), [~]Acid Volatile Sulfide (AVS), [^]other method

| STUDY YEAR | As | Cu | Cr | Pb | Zn |
|---|---|---|---|---|---|
| 1980[1^] | na | 28.5 | 25.4 | 42.6 | 107.1 |
| 1989[2^] | na | 5.7 | 6.5 | 8.3 | 18.2 |
| 1996[3~] | na | 28.7 | 53.7 | 19.8 | 127.2 |
| 1997[4#] | na | 18.6 | na | 25.5 | 75.4 |
| 2001[5^] | na | 26.3 | 25.4 | 36.1 | 107.1 |
| 2008[6*] | 1.82 | 6.49 | na | 4.89 | 29.59 |
| 2017[7+] | 8.04 | 29.5 | 51.32 | 26.01 | 104.6 |

The sediment influx to the TRE is largely terrigenous in nature, with a strong correlation to terrigenous sediment deposition from urbanized watersheds and high levels of metal concentrations in the environment (Lin et al., 2001). Knowing the source of the metals detected would be useful for land and resource managers and merits further research. Effectively identifying pollution sourcing into a system can also help improve long term planning, monitoring, and overall ecosystem health. Since terrigenous deposition is largely widespread across the southern arm and marsh platform this study suggests that long term monitoring of changes in sediment deposition, as well as metal concentrations, across the marsh platform is necessary. Comprehensive, long-term monitoring throughout the estuary will offer land managers a more complete understanding of changes in the estuary over time. Understanding both spatial and temporal changes in metal concentration of the estuary surface sediments will be effective for understanding potential ecosystem and human health implications.

### 3.6. Conclusion

Fluvial sediment deposition remains as the greatest management challenge for the Tijuana River Estuary, and this study highlights the importance of understanding sediment characteristics and metals concentration data to aid in informing management decisions. This study identifies the need for increased long-term monitoring and more in-depth metal and pollutant testing in the study area. The Southern California Bight 2008 Regional Monitoring Program which analyzed at detailed sediment chemistry for both organics and inorganics throughout the Southern California Bite Region (Schiff et al., 2011) and concluded that over the last five-year monitoring term (2003-2008) the acceptable sediment condition increased from 46% to 62% of the port/bay/harbor stratum composite. However, in estuaries these acceptable sediment conditions decreased from 86% to 62% (Schiff et al., 2011). While they were not explicitly separated for analysis, some of these grab samples were collected along the main channel in the TRE. This study revealed a sharp decrease in acceptable sediment conditions over only 5 years. This reinforces the need for detailed, long-term sediment monitoring projects specifically throughout our coastal ecosystems. We also concluded that metal concentrations were slightly greater than those of the average geogenic concentrations. Further research is needed to monitor any changes and to assess potential risk to adjacent ecosystems and urban development. These higher than geogenic concentrations of metals in our study site indicate that long term monitoring and more complete analysis of metal concentrations should be part of future research. Deeper and more complete analysis of metal concentrations using methods such as acid digestion and inductively coupled plasma mass spectrometry (ICPMS) measurements could assist with the speciation of the metals as well as assist with their bioavailability and bio-accessibility. These methods may also indicate whether there is anthropogenic influence leading to the higher metal concentrations. While elevated concentrations were detected for some metals, the levels observed do not exceed any concentrations that would deem them to be immediately hazardous. Our results are similar to those of previous studies conducted in the northern arm that indicated the TRE, when compared to other globally impacted estuaries, the TRE was less impacted by pollution from trace metals than most other estuaries (Weis, Callaway, and Gersberg, 2001). This is fortunate for the TRE for now, however, it is a relative comparison. Additional samples at input locations, coupled with planned long-term monitoring should be implemented in order to track changes in these metals over time.

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# 4. **Chapter 4:** Using geomorphic change detection and unsupervised classification from LIDAR to assess habitat distribution in the Tijuana River Estuary

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## 4.1. ABSTRACT

Coastal wetland habitats are influenced directly by subtle changes in the geomorphological make up of their landscapes. Changes to physical factors, such as elevation and tidal prism, can dictate the space available for specific habitats to thrive (Florsheim et al. 1991). This current study utilized historical LiDAR datasets (2010 and 2014) to analyze the coverage of nine wetland habitat types and detect changes in the surface elevation profile of the study site over time. The greatest change in habitat type between 2010 and 2014 was seen in the high marsh habitat zones; where the High Marsh Tidal decreased 3.49 acres and the High Marsh Non-Tidal increased 4.71 acres. Between 2010 and 2014, the marsh platform of the Tijuana River Estuary increases in elevation; accumulation is seen across the habitat zones and highest values of accumulation (20 to 35 centimeters) are seen along the periphery of the tidal channel network. This is consistent with previous studies, which document higher rates of sediment accumulation in the low marsh and salt marsh habitats, found along the tidal channels in our unsupervised classifications. While the exact values of marsh surface change include error inherent in the remote sensing process, the spatial patterns of where change occurs is consistent with historical data. Understanding these patterns of change and identifying key locations can more effectively assist land management and

restoration projects. In combination, these remote sensing techniques help to identify the areas within the study site most at risk to change.

## 4.2. INTRODUCTION

Coastal estuaries and wetlands are uniquely located at the transitional interface between the aquatic and terrestrial ecosystems, which lends itself to their high productively both economically and environmentally. Barbier et al. (2011) indicates that the overall cumulative benefits these systems produce is greater than any other single ecosystem due to their unique location. Estuarine habitats provide both these economic and environmental benefits to the communities around them (Li et al. 2018, Kirwin and Megonigal 2013). Estuarine habitats are found along coastlines around the globe; however, due to their location on densely populated coasts, they are among the most anthropogenically impacted ecosystems. Anthropogenic impacts have caused increased degradation to many of the viable ecosystem services estuaries provide, such as habitat availability and water quality. Increased urbanization can influence natural watershed processes, such as the amount of sedimentation that occurs in response to seasonal rainfall events. The watershed response to these changes lead by urbanization can dynamically alter some of the important salt marsh ecosystem functions (Biggs et al. 2015, Safran et al. 2017).

Numerous studies have identified the dominant physical features, such as elevation, salinity, and sedimentation, that dictate the structure and function of these coastal wetlands and estuaries (Barbier et al. 2011, Mitsch and Gosselink 2015, Zedler 1992). In salt marshes and coastal estuaries, many of the dominant hydrological and edaphic factors correlate with elevation; factors such as tidal inundation and duration, as well as soil moisture, salinity, and redox characteristics (Zedler et al. 1999). Due to the strong relationship with many environmental parameters, elevation-based zonation of salt marsh habitats persists as the predominant restoration model. It is largely understood that the topography and microtopography of salt marshes can alter the environment and, therefore, habitat communities, and even very small changes in elevation can alter the marsh environment (Zedler et al. 1999, Zedler 1977). Furthermore, the highest rates of deposition in salt marsh environments occur on the low elevation platforms that experience the longest durations of tidal inundation (Kirwin and Megonigal 2013). This is of particular importance for restoration and ecosystem managers due to the high numbers of endangered species that utilize these low marsh habitats. For example, in Southern California, the endangered Ridgeway Rail (*Rallus longirostirus levipes*) nests in the creek edge, low marsh vegetation (previously light-footed clapper rail) (Zedler 2011). In areas where rates of elevation change (in this case rise) are on par with local mean annual Sea Level Rise (SLR) the impact of this deposition is suspected to be lower or negligible on these tidal dependent environments; however when accretion from sedimentation elevates the marsh platform at a rate greater than SLR there is potential for increased competition between plant communities. This can lead to changes over time in the overall community make-up, examples in Southern California include low salt marsh cordgrass (*Spartina foliosa*) reducing its range and the pickleweed (*Salicornia virginica*) dominating cover and turning areas of salt marsh more monotypic in nature (Zedler 2005, Zedler et al. 1999, Zedler 2011). In addition, shifts in marsh platform elevation can leave many communities vulnerable to invasion. In fact, the world's most invasive plant species are wetland plants, making up 24% of invasions (Zedler and Kercher 2004).

Traditionally, assessments of habitat type for estuarine and coastal ecosystems have been primarily studied by wetland ecologists. More recently, the increase in use and accessibility of remote sensing technologies has allowed physical geographers and coastal geomorphologists to quantify elevation. One powerful data type utilized in studying these systems via remote sensing is Light Detection and Ranging (LiDAR). LiDAR is an active remote sensing method that measures the distance to a target by illuminating that target with a "pulsed" laser light, the reflected pulses are then measured with a sensor. The differences in laser pulse return times and wavelengths are then used to determine the distance to the target and can be used create a digital elevation model (DEM) of the target area (Lillesand et al. 2007, 6th edition). LiDAR data sets are inherently GPS connected and geo-referenced, which permits the datasets to be easily compatible with GIS systems, assisting with faster data processing. LiDAR produces a rapid pulsing of light, which allows modern systems to record more than five return pulses that can be used to identify and discriminate between features such as the bare earth and forest canopy (Lillesand 2007). The high spatial and vertical resolution of LiDAR imagery offers new data for scientists and can be utilized to better model temporal change in dynamic fluvial systems (Notebaert et al. 2008). As previously mentioned, traditionally vegetation and habitat zone survey methods are costly, time-intensive, and also invasive as they require a lot of "on the ground" time within a habitat zone. Thus, having access to better, high resolution data allows for improved observation of these systems while also spend less time in the field (Notebaert et al. 2008). Sequential LiDAR data has been shown to provide evidence of landscape change, such as illuminating spatial changes in surface features in fluvial and floodplain studies (James et al. 2009) and quantifying sedimentation rates and sediment budgets (Notebaert et al. 2008; Wheaton et al. 2009). Access to high resolution data in the form of aerial imagery, LiDAR, and multispectral satellite data can offer a new collective toolset for effective resource and habitat management.

Medium- and high-resolution aerial imagery has been utilized to better examine and understand natural processes occurring in coastal and estuarine systems. Several studies (Notebaert et al. 2008, Nelson et al. 2006, Gilvear et al. 2004, Vandenbruwaene et al. 2013) have also employed LiDAR acquired data in order to assess the impacts that urbanization in a region has had on the fluvial hydrology and channel morphology of the estuarine or riverine system. Vandenbruwaene et al. (2013) utilized both black and white aerial photographs as well as false color infrared photographs in order to extract information about changes to marsh platforms as well as map and extract information about the tidal channel networks. DEMs, created from both historical topographic surveys as well as a LiDAR (light detection and ranging), were used to extract information about the aerial extent of low to high marsh transition zones over time.

Los Peñesquitos Lagoon, located in northern San Diego County, California is a coastal estuary similar in geomorphic make-up, function, and habitat zonation to that of the Tijuana River Estuary. Greer and Stow (2003) used similar methods, combining aerial photographs and more recent digital aerial imagery in order to map habitat type conversions from 1928 to 1999. The high spatial and vertical resolutions of LiDAR imagery offer scientists data that can be utilized to better capture small physical changes and model temporal change (when used repeatedly) in dynamic fluvial and estuarine systems.

Computer automated image classification methods also offer researchers a range of benefits. They are more objective and consistent, have high repeatability, and reduce the likelihood of errors from visual classification (Tuxen et al. 2010). One example of a widely used computer automated image classification method is unsupervised classification, which is an automated pixel-based classification that involves no priority input values from the user. Unsupervised classification utilizes natural breaks and clusters within the datasets to create the automated breaks. This differs from supervised classification which requires the user to incorporate pixel training sites prior to automated classification (Tuxen et al. 2010, Klemas 2011). Both of these classification methods have been demonstrated using a variety of remotely sensed datasets in wetlands across the globe (Klemas 2011, Nelson et al. 2006). Commonly, these techniques are applied to vegetation and land-use type classifications. Belluco et al. (2006) employed unsupervised classification in addition to two other techniques (Spectral Angle Mapper and Maximum Likelihood) to five different remotely sensed data types in order to assess performance of these classifiers when applied to halophytic vegetation mapping under different conditions and with different remotely sensed data. They concluded that unsupervised classification can provide reliable classification results for halophytic marsh vegetation (Belluco et al. 2006).

Broadly, this study seeks to further assess the utility of remote sensing techniques within estuarine environments to better inform habitat management in these threatened systems. Specifically, the objectives of this study in the Tijuana Estuary are to (1) use the relationship between elevation and vegetation to evaluate the accuracy of the use of unsupervised classification to classify habitats, (2) spatially quantify habitat type change over time, and (3) identify areas that have experienced elevation changes, whether erosion or deposition. Low marsh and salt marsh habitats are critical habitats for many endangered species; if successful, ecosystem monitoring using these remote sensing techniques can provide repeated and timely data to inform restoration and land managers which habitats of interest may be subject to change.

### *4.2.1. Study Site*

The Tijuana River Estuary (TRE) is located at the southernmost extent of San Diego County in Southern California. The TRE sits at the United States-Mexico border and at the mouth of the Tijuana River Watershed, which drains approximately 1,750 mi$^2$. Three-fourths of the Tijuana River Watershed resides within Mexico and is the dominant fresh water hydrologic influence in the system. The Tijuana River Watershed has experienced a large growth in urbanization, development and changes in land use types over the last half-century (Zedler 2011, Biggs et al. 2010). The population of Tijuana, Mexico, which resides just upstream from the estuary grew from 100,000 residents in 1956 to 1,559,683 in 2010 (Biggs et al. 2010, Rodríguez and Santos 2018). This increase in population as well as increase in occupied land area has led to the construction of more impervious surfaces and elimination of once vegetated landscapes, which may increase the likelihood of sediment runoff from the watershed to neighboring regions downstream, in this case the TRE (Biggs et al. 2010). The Tijuana River as well as many ephemeral streams and sub-watersheds, such as the Goat Canyon at the southern end of the estuary, that run in to the estuary by in large accounts for the fresh water hydrologic influence in the system.

The San Diego region is dominated by a Mediterranean climate and experiences low average rainfall (<30 cm/yr) annually. However, the variability of rainfall in the region is high; and San Diego has the highest variability of streamflow in the United States, exceeding all other parts of the country for differences between wet and dry years (Zedler 2011, Zedler et al. 1986). The variability in rainfall, storm events and subsequent streamflow is predicted to increase over the next century due to impacts of climate change (Kalansky et al. 2018). Due to this Mediterranean climate, the Tijuana River is dominated by seasonal flows fueled from rainfall events, which allows for saline and tidal conditions to dominate at the lower end of the system (Cahoon et al. 1996). The TRE is a coastal plain tidal wetland and remains as the largest intact estuary in Southern California. This system has historically been home to a range of important ecosystems and habitat zones grading from low to high elevation: open water, tidal channels, low marsh/salt flat, saltmarsh, high marsh tidal and non-tidal transition zone, sage scrub, grassland, and dunes. This study focuses on the western most portion of the TRE (32º32”58.2 N and 117º07”17’.3 W) (Fig. 4.1).

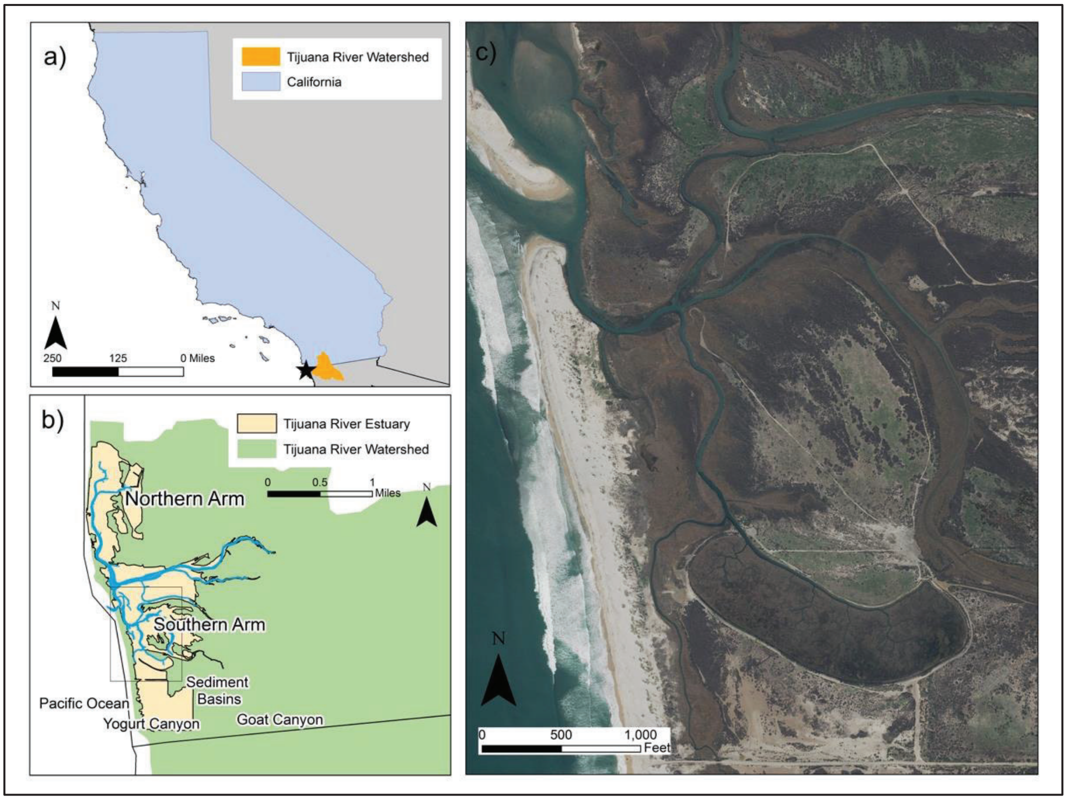

Figure 4.1. Study area: Tijuana River Estuary a) Location in southern California (32º32”58.2 N and 117º07”17’.3 W) at the mouth of the Tijuana River Watershed. b) The Tijuana River Estuary is comprised of two distinct hydrologic units, the Northern Arm which includes the Tijuana Slough National Wildlife Refuge, and the Southern Arm which includes the Model Marsh, Sediment Basins, Goat and Yogurt Canyons. c) Western portion of the TRE and focus area.

The Tijuana River Estuary has been characterized as an estuary with a reduced tidal prism and high-volume influx of terrigenous sediments (Zedler 1992, Callaway and Zedler 2004, Weis et al. 2001, Wallace et al. 2005, Florsheim et al. 1991). Wallace et al. (2005) defined fluvial sediment deposition in the TRE as the primary management challenge in this, as well as many other, southern California wetland, primarily because the sediment deposition raises the marsh surface and fills creek networks, which together reduce tidal connectivity. For the TRE specifically, studies have been conducted to understand the extent and ecological make-up of vegetation communities, as well as identify shifts in these communities from salt to fresh water tolerate species, along with the influence of invasive species (Zedler 1992, Zedler and Kercher 2004, O'Brien and Zedler 2006). While these studies provided detailed information about the vegetation make-up, they typically involved in-depth field work, such as transects, which can be costly and time intensive (Zheng et al. 2016), and therefore, infrequent. Thus, accelerated changes or changes occurring over short time periods can be missed. The time scale is important with respect to the current and sometimes rapid changes and their potential impacts on the estuary due to climate change. In the TRE, the sediment deposition on the marsh is likely to continue to exceed that of sea level rise; but it is a large consideration for restoration and habitat managers (Zedler 2011) and should be continuously monitored.

The Tijuana River National Estuarine Research Reserve (TRNERR) has monitored sediment accumulation in a few select places utilizing Sediment Elevation Tables (SETs) since 2012. The SETs employ a technique that consists of a known mineral marker, in this case a feldspar layer that are sequentially monitored for changes in reference to this point using both GPS and in field measurements. However, due to management stressors such as staff time, funding and marsh access, these SETs are not monitored regularly. The SETs are located in the northern arm of the estuary, which is hydrologically distinct from the main and southern arms. These measurements can be used more broadly to measure sedimentation in the northern arm, and are only used for relative comparison in this study.

## 4.3. MATERIALS AND METHODS

### *4.3.1. Data Sources and Preparation*

LiDAR and aerial imagery, along with metadata about the resolution, collection dates, and acquisition sources of the datasets used in this study (Table 1), were obtained from the National Oceanographic and Atmospheric Administration's (NOAA) online "Data Access Viewer", the United States Geological Survey's (USGS) online "Earth Explorer" as well as the San Diego Association of Governments (SANDAG) (NOAA 2019, USGS 2019).

Table 4.1. Spatial datasets utilized for analysis.

| Dataset | Acquisition Source | Acquisition Date | Spatial Resolution |
|---|---|---|---|
| LiDAR Point Cloud and Composite DEM | NOAA | 2010 | Vertical: 5cm Horizontal: 1.0m |
| Q1 LiDAR Point Cloud | SANDAG | 2014 | Vertical: 5cm Horizontal: 0.6m |
| Aerial Imagery | USGS | 2010 | Horizontal: 2.0m |
| Aerial Imagery | SANDAG | 2014 | Horizontal: 0.5m |

Once identified, the LiDAR point cloud datasets were downloaded and mosaicked together. We used the last return ground points (bare earth) to create a composite DEM for further analysis. This process was repeated for the 2010 and 2014 LiDAR datasets. We first clipped the mosaicked raster to include the main area of study, the southern arm of the estuary as well as the mouth to be used for the unsupervised classification and the mapping of habitat change over time, objectives 1 and 2, respectively. A secondary clip of the datasets was performed to exclude the mouth; these were used for the elevation change detection (objective 3). The aerial imagery was preferentially selected based on three criteria: (1) having the highest available spatial resolution, (2) collection and acquisition of imagery were similar to the LiDAR collection, specifically datasets that were collected in the same season, (3) the imagery selected for both 2010 and 2014 represents a similar tidal height as to avoid potential bias toward low marsh habitats. Imagery was clipped to the same spatial extent as the LiDAR datasets and 40 random points were generated to ground truth the unsupervised classification within this extent.

### *4.3.2. Habitat Classification: Unsupervised*

To assess the relationship between elevation and habitat type, this study employed unsupervised classification using ERDAS Imagine Software (Hexagon Geospatial, 2016). We determined that using nine habitat classes would provide both the broad inclusion of habitat types, while also giving us the ability to differentiate habitats influenced by tidal processes. We particularly included habitat types that are considered ecotones within the estuarine system and those that would be most subject to dramatic habitat type change (such as the transition from high marsh- tidal to high marsh non-tidal). Habitat types for this study were selected in accordance with the 2017 San Francisco Estuary Institute historical habitat study in the Tijuana River Valley (Safran et al. 2017). This study compiled a comprehensive list of habitat classification that utilized historical maps and difference vegetation indices. The nine habitat classifications utilized in the unsupervised classification for this study are as follows: Open Water, Tidal Channel, Low Marsh/Salt Flat, Salt Marsh, High Marsh (Tidal), High Marsh (Non-tidal), Sage Scrub, Grassland and Dune (Safran et al. 2017). These habitats grade from low to high elevation respectively. Habitat analysis included calculating percent cover for each of the 9 habitat classes following the unsupervised classification. This was followed by mapping the changes over time between the sequential datasets to assess potential change over time.

To evaluate the accuracy of unsupervised classification in determining habitat types within the Tijuana Estuary, we utilized high resolution aerial imagery to ground truth the results. Using ArcGIS 10.5 (ESRI, Redlands, California), we used the 40 randomly generated points to compare the habitat types from the unsupervised classification with the habitat types we identified from the aerial imagery at the location of each of point. Points that were assigned in spaces too close to the coastal shoreline and outside of the estuarine habitats were omitted, habitat zones in these locations are influenced more heavily by changes in daily tidal conditions and are not estuarine in nature. The assigned classifications were then compared against the unsupervised classification results. This allowed for a specification of which habitat types the computer most correctly identified and which it did not, as well as assess the accuracy of using only elevation data to assign habitat types. This was repeated for both sets of LiDAR data (2010 and 2014).

### *4.3.3. Change Detection*

In order to spatially quantify erosion and deposition patterns within the study area, the same two periods of available LiDAR data, 2010 and 2014, were used to perform geomorphic change detection using the raster calculator in ArcGIS 10.5. The result was a DEM of difference based on subtraction of elevations for every cell in the datasets. The mouth of the estuary was masked out of the DEMs and would not be accounted for in the change detection analysis. The study site is an intermittently open estuary and lateral migration of the mouth is natural and frequent. This lateral migration between 2010 and 2014 is visible in both the aerial imagery and DEM's and is not indicative of marsh platform sediment deposition. Due to a systematic error in the 2010 LiDAR in the eastern half of the flight path, we identified and corrected the error by clipping the eastern portion along the flight path and using raster calculator to add 10.3 centimeters (as determined by ground truthing) to the dataset. The western and eastern portions were then mosaicked back together. Due to variations in the spatial resolutions of the LiDAR between the 2010 and 2014 DEM's, both of the datasets were resampled to a 3-meter resolution prior to running the differencing analysis.

## 4.4. RESULTS

### *4.4.1. Habitat Classification: Unsupervised*

From the unsupervised classification of the 2010 and 2014 DEMs, we calculated the total percent cover by area for each of the nine assigned habitat classes, as well as the change in the total area (acres) of each of the habitat types between 2010 and 2014 (Figure 4.2, Figure 4.3, Table 4.2). The TRE has a tidal mouth subject to natural lateral migration, which and is clearly visible in the resultant unsupervised classification outputs. Throughout the last 150 years, the mouth has migrated periodically within a 1000 meter zone, as observed in historical surveys (Safran et al. 2017). Our results indicate that the largest percent area cover decrease of 3.91 acres occurred in the “Open Water” habitat class. This change can be largely attributed to the natural lateral migration of the mouth. These classification results were then used to calculate total percent cover of each of the nine habitat classes and overall changes between the datasets. The High Marsh (Tidal) habitat experienced the largest decrease in area of 3.49 acres, followed by Grassland with a decrease of 2.49 acres. The largest increase in area, 4.71 acres, occurred in the High Marsh (non-tidal). All other habitat types saw slight increases; however, they were less than half of those seen by both High Marsh habitat types.

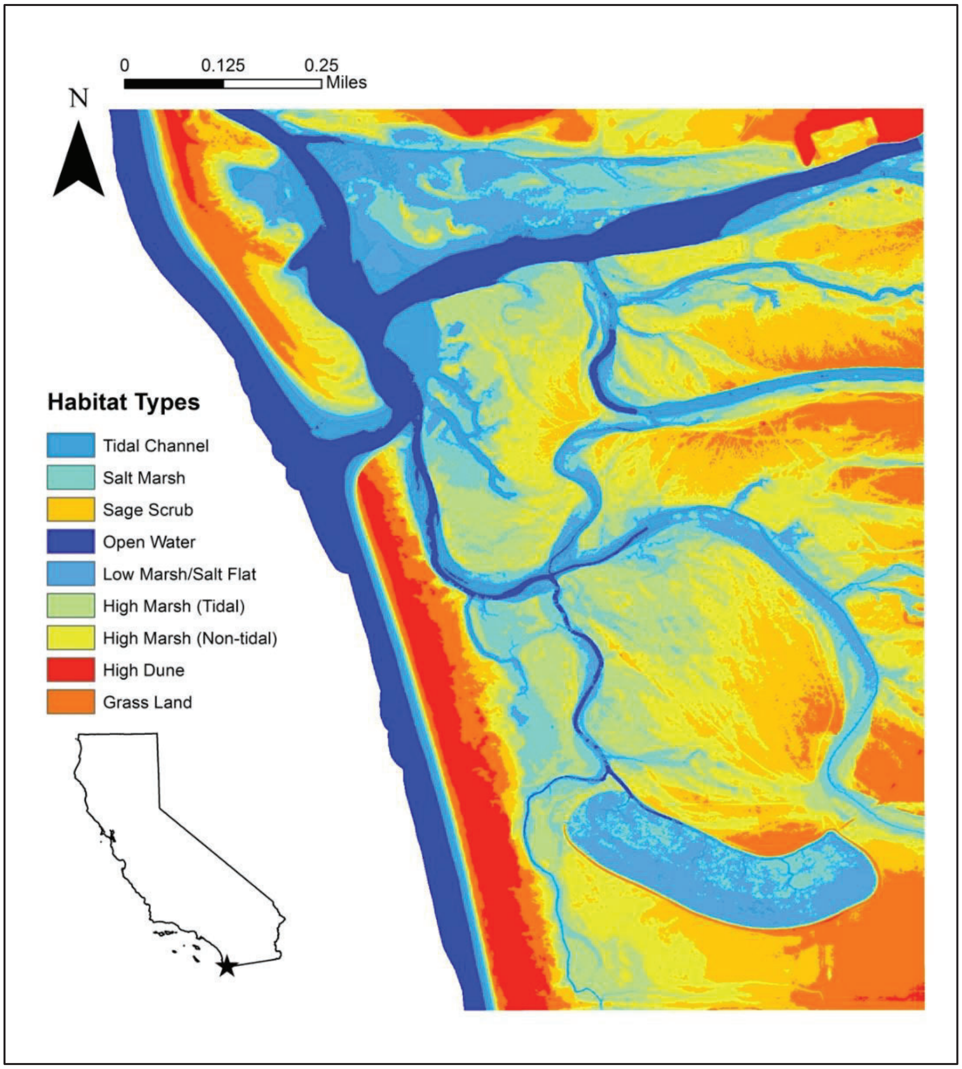

Figure 4.2. Unsupervised Classification with nine habitat types for 2010.

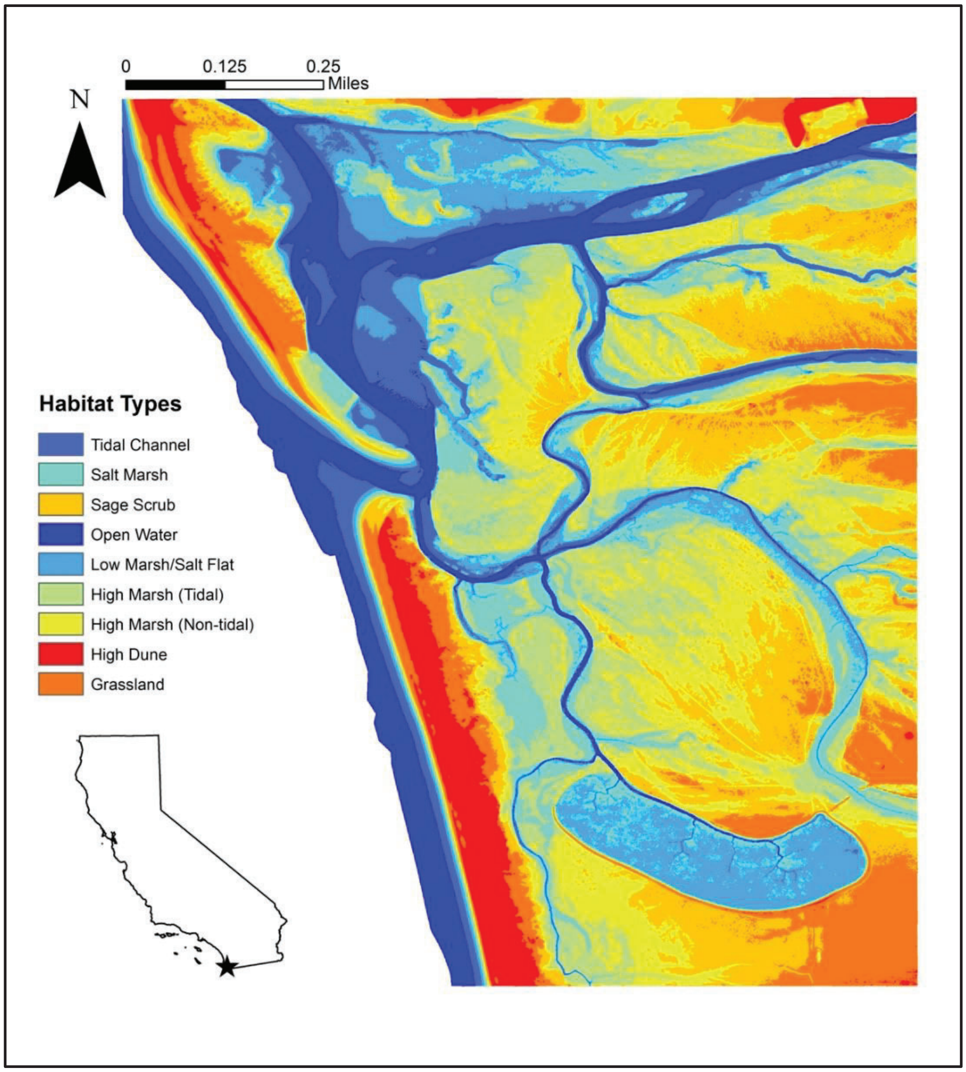

Figure 4.3. Unsupervised Classification with nine habitat types for 2014.

Table 4.2. Percent cover of the nine specified habitat classes from years 2010 and 2014 and the percent change in total acreage of habitat cover between the two datasets.

| Habitat Classification | 2010 Percent Cover | 2014 Percent Cover | Change (acres) |
|---|---|---|---|
| Open Water | 11.25 | 10.45 | -3.91 |
| Tidal Channel | 7.01 | 7.37 | 1.76 |
| Low Marsh/Salt Marsh | 10.48 | 10.64 | 0.76 |
| Salt Marsh | 9.89 | 10.16 | 1.31 |
| High Marsh (Tidal) | 14.74 | 14.02 | -3.49 |
| High Marsh (Non-tidal) | 16.67 | 17.64 | 4.71 |
| Sage Scrub | 15.30 | 15.62 | 1.55 |
| Grassland | 10.31 | 9.79 | -2.49 |
| High Dune | 4.35 | 4.31 | -0.20 |

Aerial imagery was used to assess how well the unsupervised classification of DEM was able to correctly assign habitat classes (Table 4.3). The unsupervised classification was 89% and 85% accurate in identifying habitat type for 2010 and 2014, respectively. The unsupervised classification had a difficult time discerning habitat types along the "transition zones" between habitats and if the unsupervised classification was incorrect, it always predicted a habitat type that should have been housed in a higher elevation. For example, twice in 2010 the unsupervised classification quantified a cell as "Tidal Channel" and from the imagery it was clear that habitat type was "Low Marsh/Salt Flat" (Table 4.3).

Table 4.3 Aerial imagery analysis employing 40 randomized points using ArcGIS Desktop 10.1for 2010 and 2014 datasets.

| | **2010** | | **2014** | |
|---|---|---|---|---|
| **ID** | **Unsupervised Classification** | **Imagery Classification** | **Unsupervised Classification** | **Imagery Classification** |
| 1 | High Marsh (Tidal) | **High Marsh (Non-Tidal)** | Tidal Channel | Tidal Channel |
| 2 | Tidal Channel | Tidal Channel | High Marsh (Non-Tidal) | High Marsh (Non-Tidal) |
| 3 | Salt Marsh | **High Marsh (Tidal)** | Sage Scrub | Sage Scrub |
| 4 | Salt Marsh | Salt Marsh | Low Marsh/Salt Flat | Low Marsh/Salt Flat |
| 5 | Tidal Channel | **Low Marsh/Salt Flat** | Grassland | Grassland |
| 6 | High Marsh (Non-Tidal) | High Marsh (Non-Tidal) | Sage Scrub | **High Dune** |
| 7 | Open Water | Open Water | High Marsh (Non-Tidal) | High Marsh (Non-Tidal) |
| 8 | High Dune | High Dune | Low Marsh/Salt Flat | **Salt Marsh** |
| 9 | Tidal Channel | Tidal Channel | High Dune | High Dune |
| 10 | High Marsh (Non-Tidal) | High Marsh (Non-Tidal) | High Dune | High Dune |
| 11 | Tidal Channel | **Low Marsh/Salt Flat** | High Marsh (Non-Tidal) | High Marsh (Non-Tidal) |
| 12 | Grassland | Grassland | Tidal Channel | Tidal Channel |
| 13 | Open Water | Open Water | High Marsh (Non-Tidal) | **Sage Scrub** |
| 14 | Low Marsh/Salt Flat | Low Marsh/Salt Flat | Low Marsh/Salt Flat | Low Marsh/Salt Flat |
| 15 | High Marsh (Tidal) | N/A | High Marsh (Non-Tidal) | High Marsh (Non-Tidal) |
| 16 | High Marsh (Tidal) | High Marsh (Tidal) | Salt Marsh | Salt Marsh |
| 17 | Salt Marsh | Salt Marsh | Open Water | Open Water |
| 18 | Salt Marsh | Salt Marsh | High Marsh (Non-Tidal) | High Marsh (Non-Tidal) |
| 19 | High Marsh (Non-Tidal) | High Marsh (Non-Tidal) | High Marsh (Non-Tidal) | High Marsh (Non-Tidal) |
| 20 | Low Marsh/Salt Flat | Low Marsh/Salt Flat | Grassland | Grassland |
| 21 | High Dune | High Dune | Salt Marsh | Salt Marsh |
| 22 | Low Marsh/Salt Flat | Low Marsh/Salt Flat | Open Water | Open Water |
| 23 | High Marsh (Tidal) | N/A | Salt Marsh | Salt Marsh |
| 24 | High Marsh (Tidal) | High Marsh (Tidal) | Open Water | Open Water |
| 25 | Low Marsh/Salt Flat | N/A | High Marsh (Tidal) | High Marsh (Tidal) |
| 26 | Tidal Channel | Tidal Channel | High Marsh (Tidal) | High Marsh (Tidal) |
| 27 | Low Marsh/Salt Flat | Low Marsh/Salt Flat | Open Water | Open Water |
| 28 | High Marsh (Tidal) | High Marsh (Tidal) | Sage Scrub | Sage Scrub |
| 29 | High Marsh (Tidal) | High Marsh (Tidal) | High Marsh (Tidal) | High Marsh (Tidal) |
| 30 | Tidal Channel | Tidal Channel | Grassland | **High Dune** |
| 31 | High Marsh (Tidal) | High Marsh (Tidal) | High Dune | High Dune |
| 32 | Low Marsh/Salt Flat | Low Marsh/Salt Flat | Low Marsh/Salt Flat | Low Marsh/Salt Flat |
| 33 | High Marsh (Tidal) | High Marsh (Tidal) | Low Marsh/Salt Flat | Low Marsh/Salt Flat |
| 34 | Grassland | Grassland | High Marsh (Non-Tidal) | High Marsh (Non-Tidal) |
| 35 | High Dune | High Dune | High Marsh (Non-Tidal) | **Sage Scrub** |
| 36 | High Marsh (Tidal) | High Marsh (Tidal) | High Marsh (Tidal) | High Marsh (Tidal) |
| 37 | Salt Scrub | Salt Scrub | Grassland | Grassland |
| 38 | Salt Scrub | Salt Scrub | Sage Scrub | Sage Scrub |
| 39 | Tidal Channel | Tidal Channel | High Marsh (Tidal) | **Sage Scrub** |
| 40 | Salt Scrub | Salt Scrub | Sage Scrub | Sage Scrub |

### *4.4.2. Change Detection*

A DEM of Difference (DOD) was created in order to assess the spatial variability and changes in marsh surface elevation between 2010 and 2014 (Fig. 4.4). Overall, increases in elevation can be seen across the marsh platform and of the area analyzed the largest category was between 10 and 15 centimeters of sediment accumulation on the marsh platform. Increases in elevation along the periphery of the tidal channels (red) are between 20 and 25 centimeters, these locations correspond with the low marsh and salt marsh habitats classified by the unsupervised classification (Fig. 4.2, Fig. 4.3) Areas of the study site which experienced the greatest erosion is seen along tidal channels. LiDAR has limited ability to penetrate deeply and accurately into standing water, and likely accounts for much of the variability seen along the channel edges.

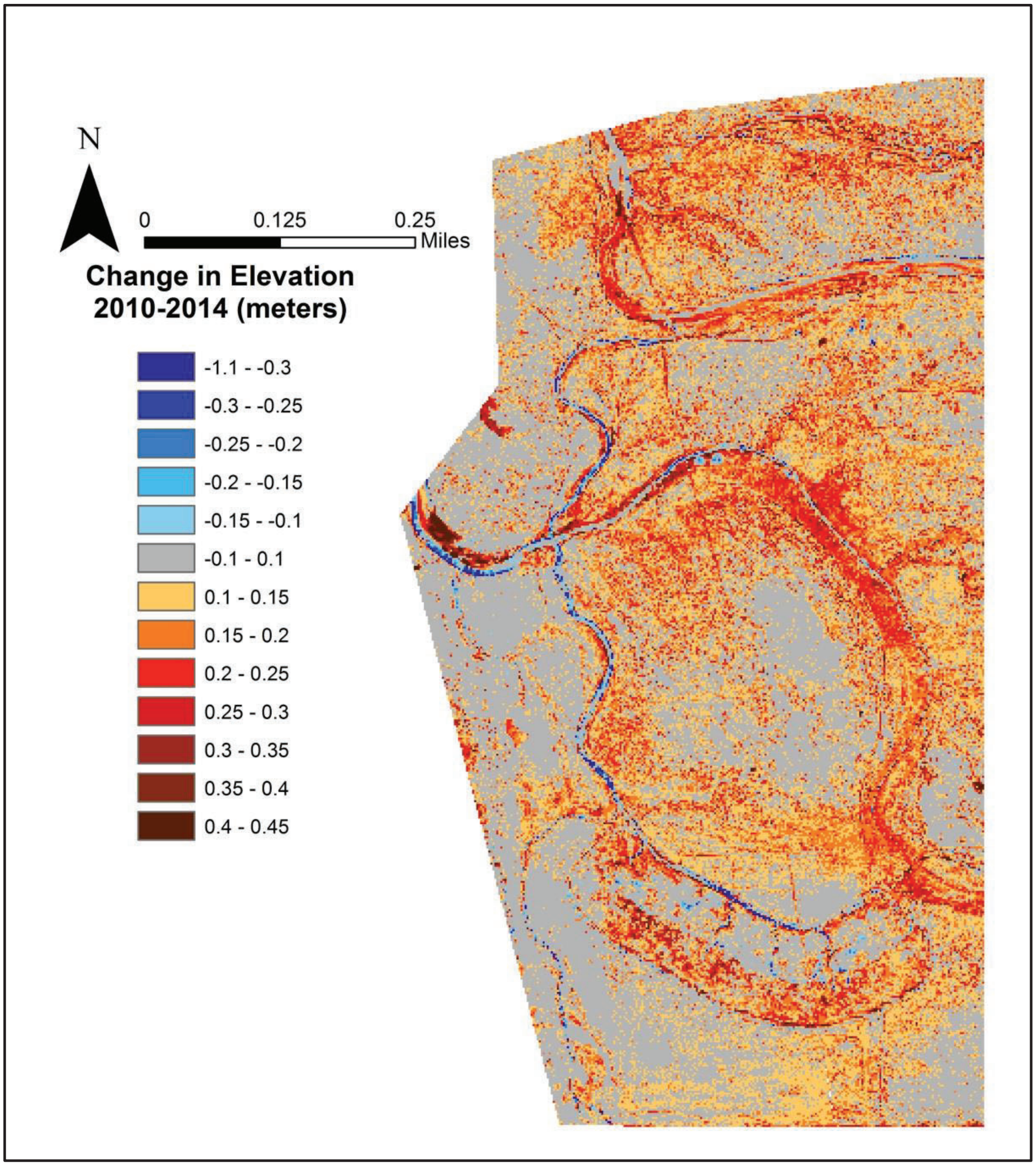

Figure 4.4. DEM of Difference (DOD) derived from LiDAR datasets from 2014 to 2010. Elevation changes along the marsh platform are in meters. The class shown in gray (-0.1 to 0.1 meters) is within the range of error associated with the resolution constraints of the data.

## 4.5. DISCUSSION

Our results show that estimating potential habitat zonation based on changes in microtopography can be accomplished with unsupervised classification. We found that unsupervised classification of the LiDAR DEM datasets was 89% and 85%

accurate for 2010 and 2014, respectively, at identifying habitat zones throughout the study area. Identifying key habitat zones in estuarine systems is critical for restoration and ecosystem management. High rates of fluvial sediment deposition continue to be the greatest management challenge in the Tijuana River Estuary (Wallace et al. 2005). Vertical accretion of sediment on the marsh platform is natural in coastal ecosystems; however, when the accretion of sediment is greater than the natural rate of sea level rise this can cause habitats to shift (Kirwin and Megonigal 2013). Accretion in the TRE is greater in areas of lower elevation throughout the marsh this is of particular interest to restoration managers because the low marsh and salt marsh habitats provide the preferential nesting grounds for multiple endangered species, such as the Ridgeway Rail (*Rallus longirostirus levipes*) (Weis et al. 2001, Safran et al. 2017, Zedler 2011).

The DOD of sequential LiDAR data identified areas that were subject to habitat type change and found that the High Marsh Tidal habitat type experienced the largest decrease in cover of 3.49 acres between 2010 and 2014. The High Marsh non-tidal habitat zone experienced the greatest increase in area cover with 4.71 acres. Zedler (2011) found that as little as a 5 centimeter increase in elevation on the marsh platform can shift vegetation communities. Furthermore, non-tidal habitats in estuarine ecosystems are more susceptible to invasive species than tidal habitats (Zedler 2011). Mapping this transition from tidal to non-tidal high marsh may identify areas of the estuary that could now be exposed to invasive species. This is important because this transition can lead to a decrease in marsh biodiversity (Zedler 2005, Zedler 2011).

Our study shows that the selected remote sensing techniques serve as a good model to highlight the spatial distribution of short-term and long-term sedimentation in the TRE. The degree of changes in elevation observed from 2010-2014, as well as where they occurred throughout the marsh platform are consistent with previous findings (Cahoon et al. 2004; Wallace et al. 2005). The periphery areas, especially along the tidal channels, saw the largest and most widespread increases in elevation, commonly between 20 and 35 centimeters of vertical change. These are also indicated by the results of unsupervised classification (Fig. 4.2, Fig. 4.3) as being dominated by the low-lying habitat zones (Low Marsh/Salt Flat and Salt Marsh). Cahoon et al. (2004) found that accretion in the low-lying salt marsh habitats in the TRE was higher than that of the high marsh; and confirmed a conservative value of 5 centimeters of sediment deposition in low marsh habitats after storm events. Wallace et al. (2005) also concluded that major sedimentation events for the TRE occurred in winters and they calculated a maximum sediment accretion of 9.5 centimeters over a six-month period. Additionally, they also found that sedimentation was highest in the low marsh areas of their study site (Wallace et al. 2005).

Southern California salt marshes and coast estuarine ecosystems present restoration managers a useful system for modeling different aspects of habitat and plant diversity and how it relates to the overall ecosystem functions. This is due to the low number of halophytic plant species (eight) and their naturally occurring community assemblages (Zedler et al. 2001, Bonin and Zedler 2008, Zedler 2005). This community structure presents an ideal model for the method of elevation-based unsupervised classification and is a good example for Southern California coastal ecosystems. In addition, degradation of the salt marsh ecosystem (increase in runoff and sedimentation) can lead to a loss in biodiversity and can lead to very large swaths of monotypic (single species) plant assemblages across the marsh (Bonin and Zedler 2008). Small shifts in elevation or changes to the marsh platform can pose threats to the native plant species; putting large monotypic assemblages at risk for invasive species (Zedler et al. 2001, Kercher and Zedler 2004).

Fluvial deposition of sediment was identified as the primary management challenge for the TRE (Wallace et al. 2005). Multiple studies have explored sedimentation rates throughout the marsh platform in response to large storm events, as well as assessing net sediment accumulation over time. Cahoon et al. (1996) quantified vertical accretion in sites throughout the northern arm of the estuary using sediment core data and found the average sediment accumulation to be 5.9 centimeters from October 1992 to March 1993. Additionally, they concluded that the higher accumulation rates correlated with the low marsh habitats, and that the pattern of sediment accumulation in the high marsh habitats differed greatly from that of the low marsh. Weis et al. (2001) calculated sediment accumulation rates using $^{137}$Cs dating and found substantially lower estimates of an average of 0.7 to 1.2 centimeters year $^{-1}$. Lastly, Wallace et al. (2005) quantified accumulation in the "Model Marsh" habitat of the TRE, which is a man-made salt marsh restoration project located within our study site. They concluded that large sedimentation events occurred during the winter storm season, with a maximum 6-month accumulation of 9.5 centimeters. In addition, they found the average accumulation from 2000-2004 for all habitats had an annual mean accumulation value of 1.3 centimeters and found that sedimentation was greatest in the low marsh habitats with an average annual accumulation of 2.2 centimeters. This pattern of lower sediment accumulation in the high marsh habitats and high sediment accumulation in the low marsh habitats corresponds with data found in our study. The DOD illustrates that throughout the marsh platform, higher values of sediment accumulation occurred along the periphery of the tidal channel network associated with the low marsh habitats (Fig 4.4). This is important, as the low marsh consists of important salt marsh and mud flat habitats that are critical for many endangered species.

The TRNERR has monitored sediment accumulation in the northern arm of the estuary since September 2012 through the use of Sediment Elevation Tables (SETs). There are four SET tables that are periodically measured, two are located in the low marsh and two in the high marsh habitats. Between September 2012 and September 2014, the SET tables in the low marsh recorded 0.76 centimeters of accumulation in the low marsh habitat and -0.16 centimeters of accumulation in the high marsh habitat. While these measurements are orders of magnitude lower than the values from the DOD using the LiDAR data, the *pattern* of higher accumulation values in the low marsh as compared to the high marsh is consistent. This highlights the need for high resolution data collection for the annual monitoring of sediment accumulation, particularly in the vulnerable low marsh habitats.

Sedimentation influenced by rainfall events have been well documented in the TRE over the last half century. The TRE's unique geographic location puts it at an increased risk climatically, as well as being subject to impacts from anthropogenic activities. Studies have shown that large rates of sediment accumulation on the marsh platform are often associated with large storm events (Weis et al. 2001, Wallace et al. 2005). The annual variability in rainfall in San Diego is higher than that of any other region in the United States; the region is impacted largely by extreme precipitation events, defined as days that have precipitation at or exceeding the 95th percentile. These extreme precipitation events account for 80% of the year-to-year variability and are therefore important to monitor (Kalansky et al. 2018, Jennings et al. 2018). In the past, these rainfall events have been coupled with large episodes of sedimentation. For example, Cahoon et al. (2004) calculated that the winter storms of 1993 mobilized approximately 5 million metric tons of sediment, and a large portion of which (31,941 metric tons) was trapped by the low salt marsh of the TRE. Webber (2010) estimates that Goat Canyon alone can deliver up to 79,000 tons of sediment to the estuary annually, and Callaway and Zedler (2004) found that up to 30 centimeters (12 inches) of sediment were deposited in the southern portion of estuary during a single (1994–1995) storm season. In the San Diego region, 2012-2015 were drought years (Kalansky et al. 2018) and exemplify the annual variability that is seen in rainfall in the region. Due to this, between our selected time frame 2010-2014, the average annual rainfall as well as the occurrence of large storm events (those that exceed the 95th percentile) was below average. This may explain why sediment accumulation rates highlighted in this study are not as substantial as those reported in the past (30 centimeters in a singular storm event) (Callaway and Zedler 2004, Weis et al. 2001, Kalansky et al. 2018).

Conversely, the 2016-2017 wet-season for the San Diego Region was considered to be an unusually wet year, access to LiDAR datasets capturing the marsh platform after this season may see larger rates of sediment accumulation (Kalansky et al. 2018). Access to high resolution sequential LiDAR data could not only help assess changes over longer time periods, but also document the potential seasonal changes and even changes caused by individual storm events, if needed. One way to acquire this data repeatedly to capture the changes associated with specific events or seasons, is through Unmanned Aerial Vehicles (UAV), now referred to as Remotely Piloted Aircraft (RPA), carrying a LiDAR sensor (Miura et al. 2018). Ultimately, this may be most useful in ecosystems where rapid and significant changes may greatly increase the risks to endangered species and their specific habitat zones.

In addition to climatic impacts, the TRE may be experiencing increased sedimentation due to land use changes in the upstream watershed. The TRE resides at the mouth of the Tijuana River Watershed, which over the past half-century has seen a large increase in population and urban development (Biggs et al. 2015, Bennett 2019). Increases in sedimentation caused by land use changes induced by urbanization corresponds with Wolman's Cycle of Erosion model (Wolman 1967). This model indicates that during the "construction" phase, land surfaces are exposed and produce orders of magnitude more sediment than the natural undisturbed surfaces would (Wolman 1967). The city of Tijuana was the fastest growing city in Mexico between 1990 and 1995 and has continued to see population growth (Biggs et al. 2015, Bennett 2019). This increased sediment runoff from urbanization in Tijuana, Mexico, has been documented by Biggs et al. (2010) with large impacts on the down-stream natural ecosystems, which is also our study site, the Tijuana River Estuary (Callaway and Zedler 2004).

Understanding how LiDAR data can aid in long-term ecosystem monitoring is useful for restoration projects. Utilizing high resolution elevation datasets for change detection of erosion and accretion, coupled with unsupervised classification of habitat types, can assist land managers in focusing resources to monitor habitat zones in the marsh that are experiencing the highest rates of elevation change. Identifying where sedimentation is occurring is the first step in determining where the sediment is coming from. Understanding these dynamics can answer important questions like why (if in excess) the sediment is not naturally being transported out of the system efficiently. In addition to understanding transport, temporal variability and migration of the mouth of the estuary may play a role in the fate and transport of sediments in and out of the estuary (Warrick et al. 2012).

Tidal prism refers to the mean volume of water moving through the systems between the mean high-tide and mean low tide. This extent has a relationship with the tidal regime and measure the inundated extent of the internal estuarine channels, basins and floodplains (Lukitina 1998, Vandenbruwaene et al. 2012, Zedler 1992). The reach, extent, and influence of tidal prism is also directly influenced by elevation changes. The TRE has long been categorized by having a low tidal prism and has since seen reductions in the past 30 years (Zedler 1992). The tidal reach and extent play a key role with elevation for dictating the habitat community structure. High resolution measurements and calculations of tidal prism are difficult and not widely available for most coastal estuaries. Using high resolution LiDAR data may also help illuminate the relationship between elevation, habitat extent, and tidal prism in coastal ecosystems.

## 4.6. CONCLUSIONS

The micro-topography of estuarine systems plays a large role in the structural make-up of the habitat zones and biodiversity of these systems. High resolution LiDAR offers a centimeter scale view of topographic changes occurring on the marsh platform, thus offering a better look at changes and shifts to the topography of the ecosystem at a micro scale; one that is suitable for assessing wetland habitat types. Understanding how to manage systems experiencing extreme sedimentation, whether due to climate change, changes upstream in the watershed, or otherwise, is a challenge. This use of high-resolution remote sensing techniques to analyze elevation changes over time enables a greater understanding of habitat areas within the estuary with the highest likelihood of change. A cell-to-cell level analysis may reveal where areas of the high marsh tidal habitat is being converted to non-tidal habitat, which can signal the potential of a further reduction in tidal prism, and an increased loss of necessary estuarine habitat. Future work is needed to continue to explore these changes in habitat type from a cell-to-cell level, as well as incorporating additional remotely sensed datasets to the classification. Sequential LiDAR datasets are effective for identifying areas within the estuary, here the Tijuana River Estuary, that are experiencing both erosion and accumulation across the marsh platform. Understanding which areas may be most susceptible to topographic changes is important for restoration and land managers for both short-term and long-term monitoring.

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# 5. Chapter 5: Summary and Conclusions

## 5.1. Summary and Conclusions

Coastal and estuarine ecosystems provide numerous benefits both environmentally and economically. They serve as the first line of defense for flood and storm control, provide important habitat space for many nesting and migratory birds, serve as a nursery for fish, and are a natural water quality and filtration systems. Economically, these systems have been shown to produce $10,000 USD per hectare annually (Barbier et al. 2011). The unique location of these systems also highlights many of the impacts they face. Located on the coast, in high human demand, they have been subject to extensive trends of increased urbanization, changes in water quality, and altered flow and sediment regimes. Many of the aforementioned benefits are crucial and highly beneficial to maintain, and land use and restoration managers employ a variety of techniques, expending significant effort and funding, to do so. Understanding many of the physical and geomorphic properties, such as sedimentation, elevation, and tidal flow, all of which dictate ecosystem form and function, is integral to effective ecosystem management.

The Tijuana River Estuary, located in Southern San Diego County, California, is a 1000 hectare coastal plain estuary system with a hydrologic regime that is influenced by semi-diurnal tides and a Mediterranean climate, with cool wet winters and warm dry summers. The estuary resides at the mouth of the Tijuana River Watershed, which extends three-fourths of its 1,750 $mi^2$ area beyond the border with the United States into Mexico, and is situated between the sister cities of San Diego, California and Tijuana, Mexico. In the last 50 years, both cities have experienced substantial urban and population growth which has put urbanization stressors on the natural watershed and these impacts are seen downstream in the TRE. One of the greatest management challenges facing the estuary is the rise in the elevation of the estuary platform from sediment deposition, occurring due to the large volume of sediments being flushed into the estuary from the neighboring southern canyons, Goat and Yogurt Canyons, during rain and large storm events (Wallace et al. 2005, Warrick et al. 2012, Weis et al. 2001).

This thesis aimed to quantify different aspects of sedimentation in the TRE with a secondary goal of contributing data to aid and improve management techniques. To do so, two distinct but related studies, with separate research questions, were conducted; the first with an emphasis on the sediment composition and distribution. The objectives of the first study were to (1) identify the characteristics of the sediment, specifically the grain size distribution and metal concentration, in the southern arm of TRE, (2) how are the metal concentrations and sediment grain size related, (3) compare metal concentrations found in TRE sediment samples to environmental screening and geogenic values. Ultimately the goal of this study is to contribute to and improve ecosystem management by identifying key areas for increased monitoring and management. Fluvial sediment deposition remains as the greatest management challenge for the TRE, and this study emphasizes the importance of understanding baseline sediment characteristics and metals data to assist in informing management decisions. This study highlights the need for increased long-term monitoring and more in-depth metal and pollutant testing in the study site. Our samples that contained high values of metal concentrations, particularly Chromium, Lead and Arsenic, were found throughout the four sub-sampling regions. Sediment input to the TRE is largely terrigenous in nature and sediment accumulation occurs across the marsh platform. Because of this, long-term monitoring can and should be implemented throughout the marsh platform. We concluded that values of metal concentrations were greater than those of the geogenic concentrations. Our study tested samples for total metal content, additional analysis is required to understand the specific chemical make-up (speciation) of the metals.

The second study in this thesis tested the utility of remote sensing techniques to assess the relationship between habitat zonation and elevation. Specifically, the objectives of the second study were to (1) demonstrate how unsupervised classification can be used to classify vegetation type using elevation, (2) quantify habitat type change over time (4-6 years), and (3) spatially identify areas within the study site that have experienced erosion or deposition over the same time period. Our research found that between 2010 and 2014, the high marsh habitats experienced the greatest amount of change in total percent area. The area of high marsh tidal habitat exhibited the greatest decrease, and the area of high marsh non-tidal habitat exhibited the greatest increase. We utilized a DEM of difference derived from LiDAR datasets between 2010 and 2014 and concluded that there was sediment deposition, of ~5–20 cm, across the marsh platform. The areas that experienced the largest accumulation were along the edges of the tidal channels. The habitats around the channel periphery from our unsupervised classification were the low marsh and salt marsh habitats. This was consistent with previous studies conducted in the TRE which indicated that the low marsh and salt marsh habitats experienced the highest rates of sediment accumulation (Wallace et al. 2005, Cahoon et al. 2004). The highest rates of accumulation (20 to 35 cm) were seen in the low marsh and salt marsh habitats. These are found, and confirmed with our unsupervised classification of habitat types, around the tidal channel network. The micro-topography of estuarine systems plays a large role in the structural make-up of the habitat zones and biodiversity of these systems. High resolution LiDAR offers a centimeter scale view of topographic changes occurring on the marsh platform, thus offering an opportunity to understand changes and shifts in the topography of the ecosystem at a micro scale. The characteristically low number of halophytic plant species and their community assemblages that make up the bulk of coastal wetlands in Southern California (Zedler et al. 2001, Bonin and Zedler 2008, Zedler 2005) make them well-suited for modeling different aspects of habitat, plant diversity, and their relationship to the overall ecosystem. Furthermore, the homogeneity of the plant community structure is ideal model for unsupervised classification (Bonin and Zedler 2008). Conversely, degradation of the salt marsh ecosystem (from increased in runoff and sedimentation) can lead to a loss in biodiversity and this can result very large areas of monotypic (single species) plant assemblages across the marsh. Small shifts in elevation or changes to the marsh platform can pose threats to the native plant species; putting large monotypic assemblages at risk for invasive species (Zedler et al. 2001, Zedler and Kercher 2004). Being able to accurately map and monitor changes in the surface elevation profile of the marsh platform can help inform management to prevent losses in biodiversity and increases in invasive species.

## 5.2. Future Work

This study adds to the data to better understand the role increased sedimentation has on the Tijuana River Estuarine system, but it also creates additional questions and new opportunities for future research. While our study assessed total metal content, we suggest a further, more extensive chemical analysis be conducted on sediment samples throughout the TRE. Further analysis quantifying the mobility and bio-accessibility of these pollutants, would be important in assessing the fate and transport of these metals and their impacts on the ecosystem. Our research suggests that long-term monitoring should include both the floodplains and tidal channels throughout the southern arm of the study area. While unsupervised classification provided an accurate method for classification of habitat make-up within the study area, further research could test and employ supervised classification methods to further refine the habitat type classification. Furthermore, future LiDAR datasets should be utilized to continue monitoring change within the system, especially with respect to continued impacts on the system via urbanization and climate change. In the San Diego County region, the 2017 winter season was considered to be anomalously wet rain year; and was accompanied by multiple high volume or 95th percentile storm events (Kalansky et al. 2018). Previous research from the study site concluded that high rates of sedimentation and accumulation on the marsh platform occurred during these events (Weis et al. 2001, Cahoon et al. 2004). Access to continued high resolution LiDAR data for the TRE after the 2017 winter season may reveal a new snapshot of both the habitat type makeup and elevation profile for the marsh platform.

Coastal estuaries and wetlands in southern California are important to the overall ecological make-up of the region. These systems serve as a front line of defense for combatting impacts associated with future projections of Sea Level Rise from climate change as well as flooding from increased variability in extreme storm events (Kalansky et al. 2018). In addition to providing coastal protection, these systems serve as the last habitat stands for many federal and state protected species. Increases in urbanization throughout the last century have imposed pressures and caused detrimental degradation for over 90% of Southern California's coastal ecosystems. Urban infrastructure has restricted their spatial extent and natural mobility to retreat. Effective and efficient maintenance of these systems is critical for future ecosystem health and this research builds upon improving that methodology.

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**APPENDIX A:** Sediment Grain Size along the channel cross sections. Sample distance indicates the location along the cross section (meters) sampled.

| Cross Section | Sample Distance | D50 Value |
|---|---|---|
| | 10m | 9.28 |
| | 1m | 26.10 |
| | 25m | 8.55 |
| | 32m | 10.37 |
| | 40m | 10.90 |
| | 42m | 16.89 |
| | 44m | 12.34 |
| | 48m | 10.96 |
| East 1 | 53m | 39.83 |
| | 0m | 31.51 |
| | 1m | 9.23 |
| | 36m | 10.30 |
| | 54m | 15.22 |
| | 55m | 29.50 |
| East 2 | 57m | 75.77 |
| | 12m | 20.62 |
| | 13m | 49.38 |
| | 16.5m | 20.38 |
| | 33.5m | 13.89 |
| | 42.5m | 8.94 |
| | 45.5m | 45.24 |
| East 3 | 18m | 26.53 |
| | 12.5m | 62.96 |
| | 16m | 46.46 |
| | 1m | 35.07 |
| | 23m | 19.86 |
| | 3.5m | 35.01 |
| | 6.5m | 15.33 |
| Main 1 | 9m | 91.07 |
| | 0m | 12.95 |
| | 12m | 15.08 |
| | 2.5m | 10.91 |
| | 21m | 9.22 |
| | 26m | 9.30 |
| | 32m | 59.29 |
| | 6m | 34.73 |
| Main 2 | 9m | 50.42 |
| | 0m | 46.78 |
| | 12m | 7.96 |
| | 20m | 8.71 |
| Main 3 | 24m | 80.27 |

|  |  |  |
|---|---|---|
|  | 27.5m | 200.07 |
|  | 31m | 65.74 |
|  | 33m | 17.85 |
|  | 36.5m | 8.84 |
| South 1 | 0m | 45.62 |
|  | 10.5m | 59.40 |
|  | 3m | 83.02 |
|  | 5m | 195.25 |
|  | 8m | 20.79 |
| South 2 | 0m | 12.91 |
|  | 11m | 12.01 |
|  | 2m | 34.47 |
|  | 5m | 670.15 |
|  | 8m | 29.37 |
| South 3 | 0m | 11.80 |
|  | 2.5m | 25.36 |
|  | 4m | 48.38 |
|  | 5m | 15.61 |
|  | 8m | 17.73 |
| South 4 | 0m | 28.48 |
|  | 11m | 19.05 |
|  | 14m | 33.90 |
|  | 16m | 46.01 |
|  | 17.5m | 19.06 |
|  | 23m | 443.58 |
|  | 5m | 19.82 |
| North 1 | 0m | 497.65 |
|  | 11.5m | 9.50 |
|  | 16m | 542.31 |
|  | 18.5m | 597.94 |
|  | 22m | 43.01 |
|  | 25m | 9.36 |
| North 2 | 3m | 60.45 |
|  | 7m | 24.43 |
|  | 10.5m | 25.50 |
|  | 13m | 19.89 |
|  | 17m | 14.07 |