Sea Level Rise Impacts to Coastal Habitats in Southern California Estuaries

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Executive Summary

The response of Southern California's coastal systems to ongoing climate change is highly uncertain. Because these systems comprise a network of economically and ecologically linked resources, a unified effort is needed at a regional level to properly restore and manage these systems into the future. To inform such an effort, additional insights into sea level rise (SLR) impacts are needed at scales that are meaningful to both site managers and regional regulators.

This project aims to improve our understanding of regional SLR impacts by assessing site-specific habitat change in response to SLR. For the purpose of this effort, we developed a habitat change index (HCI) that is used to describe the amount of divergence a system is expected to undergo from its current habitat composition. Several regionally important drivers were used in order to predict how the elevation and inundation within each system may change in the future. Drivers include sea level rise, marsh accretion and estuarine mouth dynamics (Figure ES1). Our model allows us to predict future changes in elevation and inundation, and the resultant impacts to habitat composition. The summation of site-specific habitat change provides a high-resolution, synoptic understanding of regional SLR impacts.

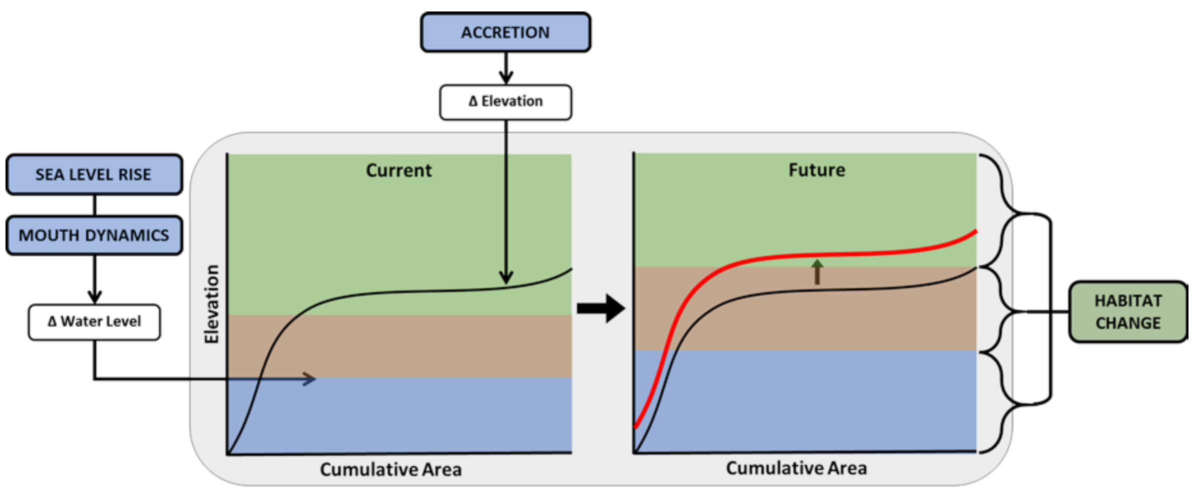

Figure ES1. The conceptual model for the habitat change assessment showing the drivers and responses that are important in determining SLR response in the Southern California Region.

Our assessment was conducted for 104 systems throughout the Southern California Bight (SCB) which range in size, structure and initial habitat composition. Overall, we found that for the 2050 SLR scenario, 12% of vegetated marsh and flats will be lost. For the 2100 SLR scenario, the percent of lost vegetated marsh and flats increased to 48% (Figure ES2). However, allowing wetlands to migrate into adjacent transitional areas could create up to 160% more wetland areas by 2050. SLR impacts and migration opportunities vary substantially by estuary archetypes; our findings suggest that fragmented river valleys are the most susceptible to SLR-induced habitat change. There are additional management actions, such as augmenting accretion and managing estuary mouth state, that may alleviate the negative impacts to habitat composition predicted in our SLR response assessment.

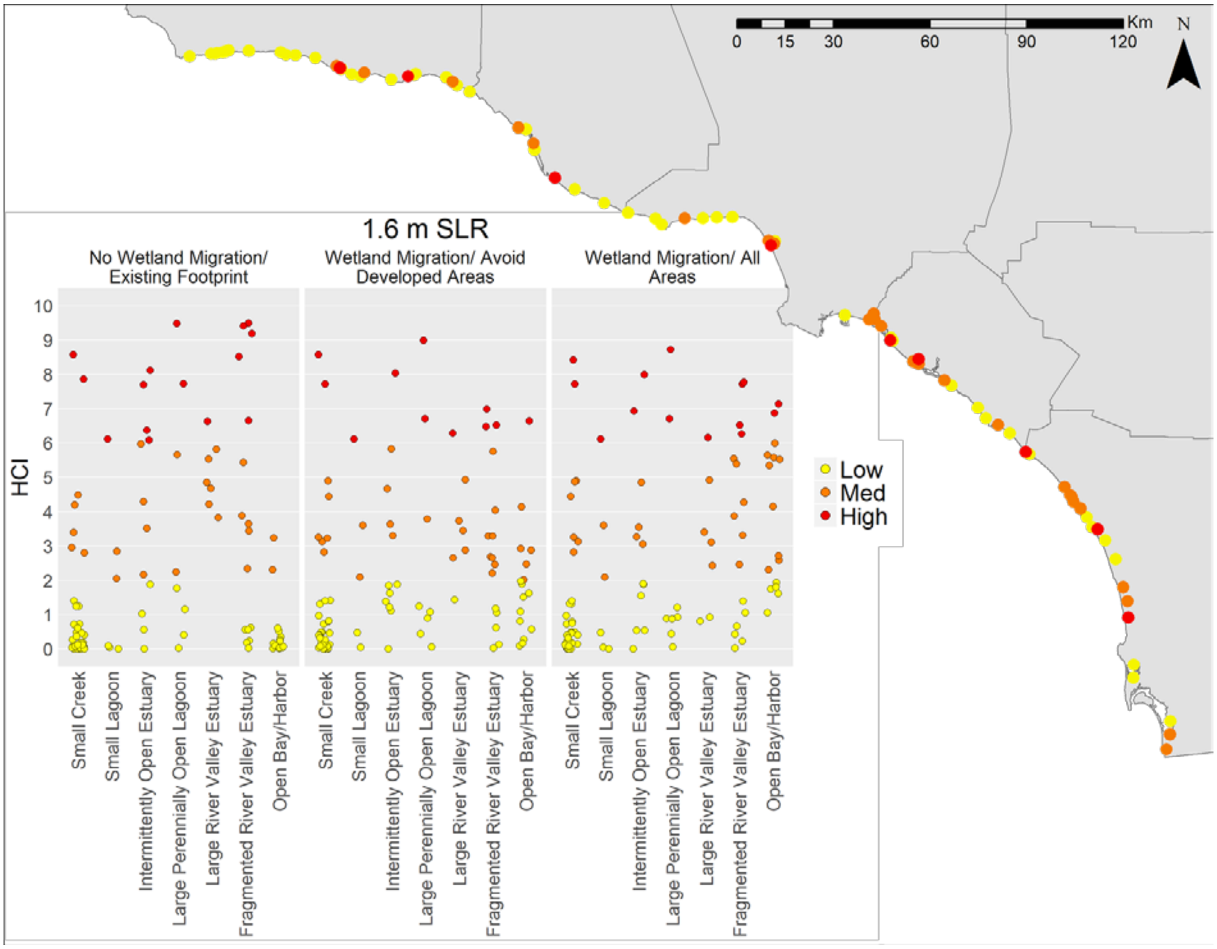

Figure ES2. 2100 SLR Habitat Change Index (HCI) for the 104 study sites throughout the Southern California region.

The work presented here is part of an ongoing effort to ensure the resiliency of coastal systems by establishing unified regional goals and management strategies. The project collaborative is known as the Southern California Wetlands Recovery Project (SCWRP), which consists of 18 agencies, including academic and governmental research institutions, regulatory agencies, and non-profit organizations, whose roles include coastal research, management, and regulation. Our findings will aid the SCWRP directly by providing insight on what systems are vulnerable to SLR. The results of our analysis will inform where additional efforts are needed to mitigate negative impacts and where there are data gaps in our current understanding of the factors contributing to SLR response. Our regional habitat change model provides a screening level assessment that can be used to prioritize and support more detailed site-specific investigations, and will provide a platform to prioritize future work based on greatest vulnerabilities or uncertainties.

Background

Coastal systems in Southern California are threatened by a number of anthropogenic and climate change impacts, which render the fate of these systems highly uncertain (PRBO Conservation Science 2011). Historical losses of up to 75% since the 1800's have created a network of coastal system remnants which are geographically sparse and limited in extent along the coast (Zedler 1996, SCWRP 2001, Stein et al. 2014). "Coastal squeeze" is prevalent in this region, which is the result of human infrastructure constraining the extent of coastal systems (SCWRP 2001, Torio and Chmura 2013). In addition, these systems are at the precarious land-sea interface and face an onslaught of climate change impacts, such as drought, storm events and most notably, sea level rise (SLR) (Scavia et al. 2002). Because these systems are highly susceptible to anthropogenic and climate change impacts, there is a need for a unified regional strategy in order to successfully restore and preserve these valuable systems (Zedler 1996). Such an approach would integrate site-level and regional management in order to create feasible goals, allow for tradeoffs between systems, and reduce redundancy and cost. Integrated planning and management efforts are currently being developed in the Southern California region by a group of agencies known collectively as the Southern California Wetlands Recovery Project (SCWRP). This group is overseeing the effort to update a regional restoration strategy to ensure the resilience of coastal systems in Southern California.

To inform such a regional effort, we must also improve our understanding of regional response to sea level rise (Sutula et al. 2006, Stein et al. 2014). SLR effects are currently one of the most challenging climate change impacts for coastal wetland restoration due to the many uncertainties associated with SLR projections, as well as marsh response to SLR. Along the coast of California, sea level has risen approximately 20 cm over the past century and is predicted to increase from 1.0 – 1.4 m by 2100 (National Research Council 2012). Changes to sea level may have enormous implications for coastal planning, land use and development, and pose an increased risk of flooding, inundation and storm hazards to coastal communities (Heberger et al. 2011, National Research Council 2012). In addition to socioeconomic impacts, coastal areas face negative ecological impacts to important systems like bays, estuaries, and lagoons, because their response to SLR remains highly uncertain.

Predicting wetland response is challenging given the complex processes at work in coastal areas. In general, wetland response is determined by the ability of a system to keep pace with SLR, which results in the persistence of systems that can gain elevation (less vulnerable) and the loss of systems that lose elevation (more vulnerable). The ability of wetlands to maintain elevation is driven by a complex interaction of environmental drivers, the geophysical response to these drivers, and the biological response of marshes to both environmental and geophysical factors (Reed 1990, Day et al. 2008, Kirwan and Megonigal 2013). For example, marsh elevation can by influenced by large-scale processes of vertical land motion arising from plate tectonics (Wöppelmann and Marcos 2016), local processes such as compaction or subsidence (Mayuga and Allen 1969, Takekawa et al. 2013), the availability of fluvial sediment discharge (Weston 2014), and biological marsh accretion (Swanson et al. 2014). As these processes contribute to marsh elevation, an additional set of processes are influencing inundation levels. In addition to the increased inundation associated with SLR, water levels in coastal systems are also impacted by interdecadal climate oscillations (Meltzer, Unpubl. Data) and estuary mouth dynamics, which are largely controlled by the interaction of fluvial inputs and marine processes (Behrens et al. 2015). These factors must also be considered when predicting marsh response to SLR and regional vulnerability.

Marsh response to SLR has been investigated in several case studies in the Southern California region. The work of Thorne et al. (2016) provides insights to SLR impacts for several sites in in the region using intensive field surveys and the Wetland Accretion Rate Model of Ecosystem Resilience (WARMER). In addition, there are several web-based viewers which display the results of SLR impact analyses in CA. For example, the Nature Conservancy's Coastal Resilience tool shows marsh migration with SLR for Mugu Lagoon in Ventura County (coastalresilience.org). While these examples provide valuable predictions of SLR response in this region, these results are limited to large, well-studied sites and may not be applicable to the wide variety of coastal systems in this region. Coastal system present in Southern California range from expansive open bays, such as San Diego Bay, to small coastal creeks like those in the Santa Ynez coastal range. As part of the SCWRP effort, these systems have been mapped for the entire Southern California Bight. Overall there are approximately 104 systems, and the majority of these are understudied, lacking data, and unpredictable in terms of SLR response. Because the response of individual systems remains highly uncertain, we therefore know little about how the region as a whole will response to SLR. Recent improvements to regional vulnerability assessments have gone beyond environmental drivers and geomorphological characteristics to include the dynamic responses of coastal systems (Lentz et al. 2016). The work of Lent et al. 2016 provides a more nuanced depiction of SLR vulnerability that indicates which types of coastal systems have the capacity to dynamically respond to SLR and which may become submerged. Both vulnerability assessments and SLR response models highlight the need for a screening level tool that can be applied to a broad geographic region, but provide insights at local spatial scales.

We addressed this need by developing a method which allows for the assessment of SLR impacts to individual systems and can be applied to all 104 wetlands in the region. In addition, we wanted to address the high spatial variability in marsh response by parameterizing our model with site-specific data and using regional data as defaults when these were not available. Understanding how each system will respond to SLR is necessary in order to determine resiliency at a synoptic scale and to inform regional planning. Here, we present an approach based on wetland typology that leverages existing data and models and uses basic relationships to estimate potential changes associated with SLR to wetlands across the region. Our goal in this assessment is to estimate relative habitat change within the region, which we quantified as the potential change in habitat composition for 2050 and 2100 for all 104 systems currently included in the RSU project. In addition, we modeled habitat change for 3 scenarios of wetland expansion: 1) No Migration/Existing Footprint – the future extent of marshes is restricted to the current (2015) marsh boundaries; 2) Wetland Migration/Avoid Developed Areas – marshes can expand in the future into adjacent undeveloped areas; and 3) Wetland Migration/All Areas - marshes can expand in the future into all adjacent areas based on their elevation, regardless of the current land use. We also investigated 4 additional management actions: 1) Increasing accretion rates; 2) Implementing one-time thin-layer sediment augmentation; 3) Managing estuary mouth dynamics; and 4) Reconnecting fragmented systems. Our findings can be used to support regional planning efforts by providing a regional screening level assessment of SLR impacts to habitat composition. Ultimately, this analysis will aid the SCWRP in prioritizing management strategies that mitigate the effects of sea level rise and thus ensure the future resiliency of Southern California coastal systems.

Methods

Model Overview

We assess potential changes in habitat composition by modeling the combined effect of sea level rise, accretion, and changes in estuary water levels due to mouth dynamics. Together, these effects alter the hypsometry, or the measurement of land elevation relative to the tide, of each of the 104 wetland systems in the region (Figure 1). Changes in hypsometry and estuary water levels impact the distributions of wetland habitats, which are generally defined by inundation frequency and elevation (see Appendix 1 for details). We ran the model for two conservative future sea level rise scenarios, 0.6 m cumulative sea level rise (currently projected around 2050) and 1.6 m cumulative sea level rise (currently projected around 2100). In addition, we predict future habitat compositions for three scenarios of wetland expansion: 1) No Migration/Existing Footprint – the future extent of marshes is restricted to the current (2015) marsh boundaries; 2) Wetland Migration/Avoid Developed Areas – marshes can expand in the future into adjacent undeveloped areas; and 3) Wetland Migration/All Areas - marshes can expand in the future into all adjacent areas based on their elevation, regardless of the current land use.

Model components are discussed in detail below, including how data were collected, processed and used in our habitat change assessment. An overview of data inputs is shown in Table 1.

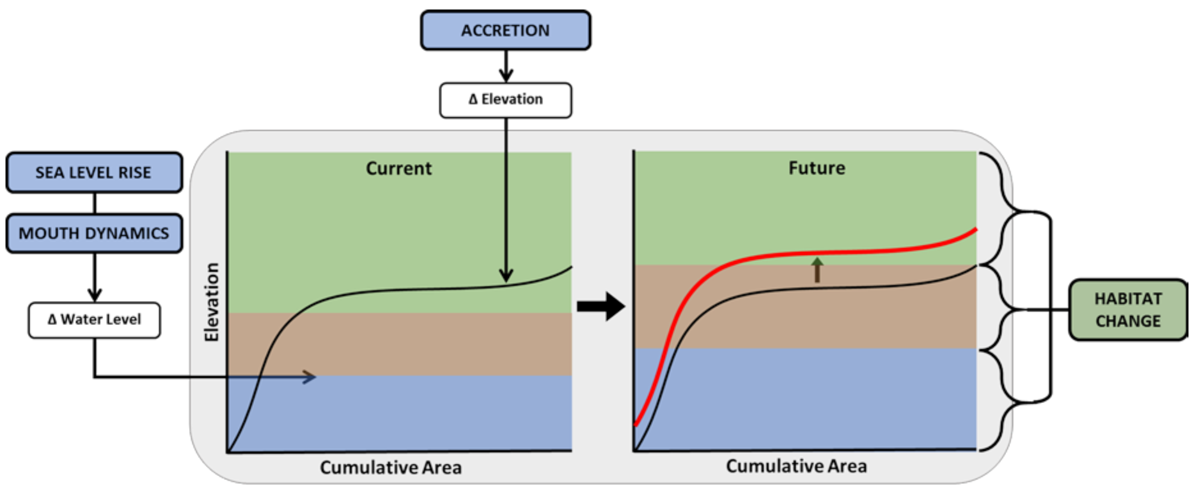

Figure 1. The conceptual model for our modeling framework. Model inputs (blue) are used to calculate changes in water levels and elevation. Water level changes alter the marsh zones defined by elevation (background colors indicating subtidal (blue), mudflat (brown), and marsh zones (green)). Elevation changes due to accretion raise the current marsh hypsometry (black line) to future hypsometry (red line) relative to the current tidal datum. Habitat change is the difference in area under the curve for each marsh zone under current (black line) and future (red line) conditions.

Table 1. Data inputs for the SLR-induced habitat change model.

Input	Scale	Value (units)	Source
Relative Sea Level Rise			
SLR 2050 Projection	Regional	12.2 (mm yr^{-1})	NRC 2012
SLR 2100 Projection	Regional	16.6 (mm yr^{-1})	NRC 2012
VLM 2100 Projection	Regional	-1.5 ± 1.3 (mm yr^{-1})	NRC 2012
t_0, t_1, t_2		2016, 2050, 2100 (yr)	
Accretion			
Measured Accretion ± Error	Site	(mm yr^{-1})	SCCWRP Literature Review
Mouth Dynamics			
Daily Water Levels	Sub-regional	(m)	NOAA
Daily Wave Height, Period, Direction	Sub-regional	(m, s, degrees)	CDIP
Watershed Run-off Estimates	Sub-regional	(m^3 s^{-1})	SCCWRP
Estuary Mouth Width	Site	(m)	SCCWRP
Estuary Closure Estimates	Site	(%)	SCCWRP
Estuary Area	Site	(km^2)	SCCWRP
Habitat Change			
Current Habitat Extent	Site	(m^2)	SFEI/SCCWRP
Habitat Elevation Relationships	Sub-regional		SCCWRP/SFEI Literature Review
Estuary Hypsometry	Site		SFEI/SCCWRP

Archetype Classification

In Southern California, coastal systems can be categorized into several classes based on geomorphology, mouth dynamics, size and plant species composition (Jacobs et al. 2011, Stein et al. 2014). The approach used to develop these classes and to assign individual wetlands to these classes has been described elsewhere (RSU Archetype chapter). The resulting classes are commonly referred to as "archetypes", which include the following: 1) small creek systems; 2) small lagoons; 3) intermittently closer river mouth estuaries; 4) large perennially-open lagoons; 5) large, depositional river valleys; 6) fragmented marsh remnants; and 7) open harbors and bays (Stein, RSU Archetype Chapter) (Table 2). Closed or intermittently closed lagoons represent a large component of coastal systems in Southern California (Zedler 1996, Clark et al. 2013). These systems are characterized by periodic formation of bars at the estuary mouth when precipitation and fluvial inflow are too low in areas of high coastal exposure (Roy et al. 2001, Potter et al. 2010). Each of the 104 focal systems of this study has been assigned to one of these seven archetypes.

Archetype groupings represent systems with similar physical structure and ecosystem drivers that are expected to react in similar ways to sea level rise. The regional strategy uses the concept of archetypes to facilitate the extrapolation of information between systems, from data-rich sites to those less studied, in order to fill the gaps in our knowledge of the region. Where applicable, we used data aggregated by archetype to provide model inputs for systems lacking site-specific data (see sections *Accretion* and *Mouth Dynamics* below for more detailed examples). Using the archetype framework to extrapolate input data, we were then able to interpret SLR response for all 104 systems and the region as a whole. In addition, the archetype classes help the regional strategy in organizing the development of the regional objectives.

Table 2. Archetype classification of coastal systems in Southern California. Adapted from RSU Archetype Chapter.

Code	Name	General Description	Associated Habitats
1	Small Creek	Small creek systems; minimal subtidal habitat area; generally higher gradient	Intertidal (Cowardin), Riparian marsh and meadow (calveg)
2	Small Lagoon	Small coastal lagoon without an associated creek	Intertidal and subtidal habitats. May have fringing riparian marsh
3	Intermittently Open Estuary	Intermittently closing river mouth estuaries	Intertidal (Cowardin), Riparian marsh and meadow (calveg)
4	Large Perennially-Open Lagoon	Open basin, extensive subtidal habitat, fringing intertidal;	Intertidal emergent, pickleweed and/or cordgrass habitats (calveg)
5	Large River Valley Estuary	Large, depositional river valleys, fringing marsh; high dynamic ratio	Intertidal emergent, pickleweed and/or cordgrass habitats (calveg), moderate subtidal area (Cowardin)
6	Fragmented River Valley Estuary	Currently fragmented large depositional river valley; opportunities for reconnection	Intertidal emergent, pickleweed and/or cordgrass habitats (calveg), moderate subtidal area (Cowardin)
7	Open Bay/Harbor	Open water harbors, bays, lagoons; large area, wide & low-lying mouth	Dominated by subtidal habitat

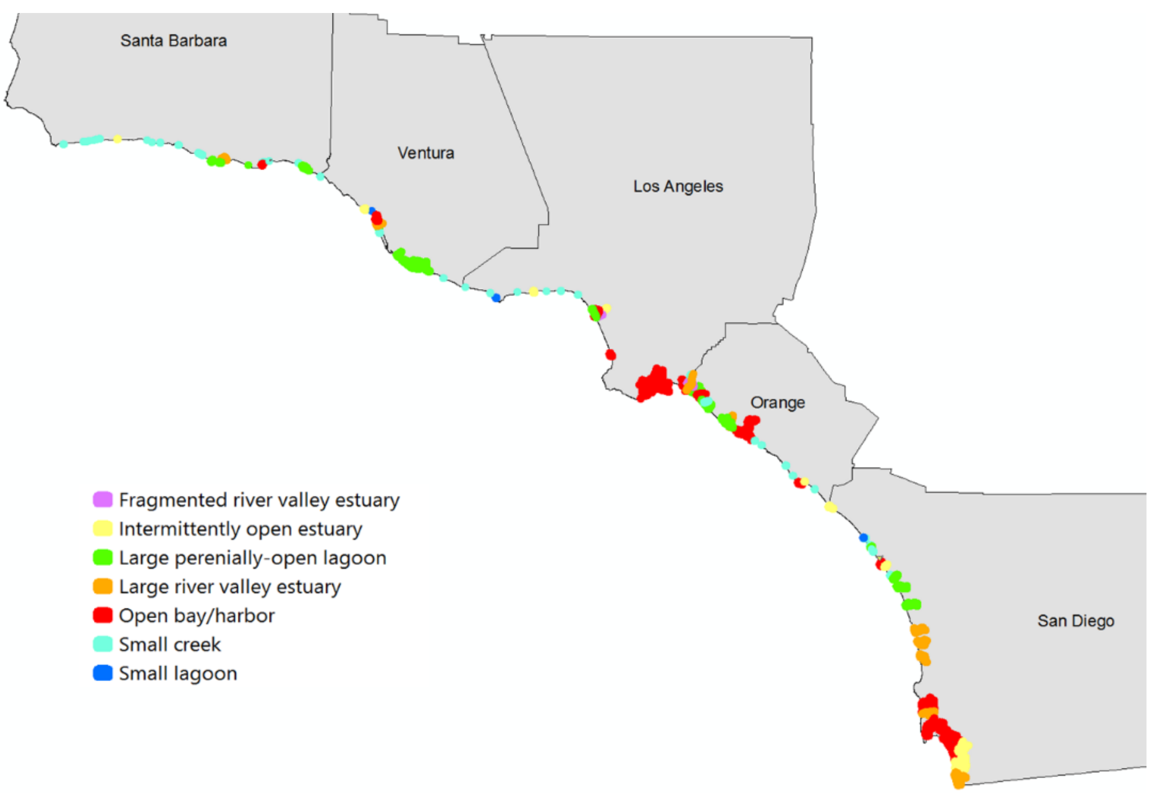

Figure 2. Focal estuaries in the Southern California region. Symbology indicates archetypes or groupings of estuaries based on similar characteristics. Estuaries sizes are expanded for visualization and are not to scale.

Model Components

Relative Sea Level Rise (rSLR)

We used the projections provided by the National Research Council's Report on *Sea Level Rise for the Coasts of California, Oregon and Washington: Past, Present and Future* (2012) to inform our regional estimates of SLR and vertical land motion (VLM). The committee's SLR projections incorporate contributions from steric and dynamic ocean, glaciers and ice caps, and vertical land motion (National Research Council 2012). Sea level projections for the Los Angeles area range from 12.7 - 60.8 cm for the year 2050, and from 44.2 – 166.5 cm for 2100. Because these estimates are likely conservative, we selected the maximum of SLR projection ranges for 2050 and 2100 (60.8 cm and 166.5 cm, respectively). We converted these projected levels of inundation (cm) to SLR rates (mm yr^{-1}) by dividing by the difference in time period between our modeling time points (2050, 2100) and the NRC 2012 baseline (2000). As a result, the SLR rates used in our assessment are 12.2. mm yr^{-1} for 2050 and 16.6. mm yr^{-1} for 2100.

NRC 2012 projections for the California coast south of Cape Mendocino include a VLM contribution of -1.5 ± 1.3 mm yr^{-1}. Because VLM is already accounted for in the NRC 2012 SLR projections, we did not include VLM in our model calculation of relative SLR (rSLR). However, we wanted to preserve VLM as an important component of the model to allow for future inclusion of site-specific data. For example, local tectonics, compaction, liquid extraction and fluid recharge are important factors in producing locally high rates of VLM (National Research Council 2012). In Southern California, shallow subsidence has been linked to anthropogenic groundwater and oil extraction in the Los Angeles area (Mayuga and Allen 1969, Bawden et al. 2001, Argus 2005). Although local, site-specific estimates of subsidence are not currently included, VLM has been preserved as a model component because we plan to improve this in the future, but also to allow users to parameterize the model when local expert knowledge is available.

Accretion

Empirical estimates of accretion were obtained through a review of published literature pertaining to coastal CA systems. Our goal was to obtain records of site-specific field measurements of accretion for as many of the 104 systems as possible. Relevant information noted with each record of accretion included the associated estuary, time period, methodology, a description of the reported accretion type (e.g., long-term, short-term, net, organic, mineral, storm deposition) and if applicable, the marsh zone (e.g., low-, mid-, high-marsh) where accretion was measured. Because of the wide range of methods and unit measurements reported in the literature, accretion estimates were standardized to mm yr^{-1}. We excluded records of short term (< 10 yr) accretion and sediment deposition resulting from episodic storm events. This ensured that the accretion estimates used in our model reflect long-term marsh accretion patterns occurring at a scale comparable to SLR projections. The model did not account for fluvial runoff or mouth dynamics, which may influence future accretion rates.

Because of the limited availability of published accretion records specific to this region (See section *Accretion* in *Results* for more details), we used the archetype framework to extrapolate data from well-studied systems to data-poor systems. When empirical data was available for a given system, we used site-wide averages across all marsh zones as accretion input. For systems lacking empirical data, we aggregated accretion estimates from our literature review by archetype to provide realistic inputs for similar systems specific to this region.

Mouth Dynamics

We included an estuary mouth dynamics component in our assessment due to the high prevalence of intermittently opening and closing estuaries in this region. Changing mouth state is a product of marine and fluvial drivers, such as wave energy and river discharge, which contribute to changes in estuary water levels by altering accretion and tidal prism. The convergence of marine and terrestrial inputs, as well as the resulting mouth dynamics, play a role in the evolution of coastal systems under accelerated sea level rise. It is likely that SLR will increase the frequency of estuary closure, inducing a shift in dominant mouth state, and will ultimately amplify water levels within the system to alter habitat composition. Previous SLR response assessments have not included mouth dynamics in determining marsh response to SLR.

The mouth dynamics component of this modeling effort was applicable to most archetypes in this region, excluding the large open bays/harbors and systems with mouths that have been engineered open. Our rationale for excluding engineered systems uses local, expert knowledge indicating that these systems are continually managed to remain open. This applies to all the large open bays and basins (archetype 7) in the region, but also to a number of other systems classified as archetypes 3, 4, 5 and 6. In total, 35 of the 104 systems have engineered mouths.

We created a simple model to evaluate the potential impacts that SLR may have on estuary mouth state and the subsequent changes to lagoonal water levels. We created a synthetic daily time series for current, 2050 and 2100 sea levels using local NOAA tide level data (tidesandcurrents.noaa.gov), CDIP water level and wave data (cdip.ucsd.edu), and Southern California Coastal Water Research Project (SCCWRP) coastal watershed run-off data. To predict "future" closure indices, we manipulated wave and tidal inputs to reflect sea level rise increases for 2050 and 2100 based on NRC 2012 projections. Data was sufficient to create synthetic time series for our mouth dynamics model for 36 out of the 104 systems.

In order to estimate the probability of changing mouth state, we used the closure index (S) metric used by Williams and Cuffe (1995) and PWA (2003):

$$S = P_w/P_t$$

$$P_w = 0.5\rho g H_s C$$

$$P_t = (\gamma h_T)/b * (\Omega/T + Q)$$

Where P_w is wave power, ρ is the constant 1 kg L^{-1} for water density, g is the constant 9.81 m s^2 for acceleration by gravity, H_s is significant wave height and C is wave group velocity (C=1.56*s, where s is wave period). Tidal power (P_t) is also described above, where γ is the constant 1000 kg unit weight of water, h_T is the tidal range, b is the estuary breach or mouth width, Ω is tidal prism ($\Omega = h_T*A$, where A is the water surface area of the basin), T is the ebb tide period and Q is fluvial discharge. We estimated the daily likelihood of closure for each estuary over the entire time series.

Next, we estimated how water levels may change in a given system when it is predicted to be open ($S < 0.1$) or closed ($S > 0.1$). In doing so, we made the assumption that every time closure risk is above the threshold, the system closes. This over-predicts mouth closure, however, we wanted to calculate the percent increase in predicted closure associated with SLR and the hypothetical changes to lagoonal water level based on the time series data. Also, this mouth dynamics response threshold allowed us to repeat this process and compare outcomes for all 36 systems where data was available. With this in mind we estimated daily lagoonal water levels for systems with both "open" and "closed" conditions using a simplified model based on the work of Behrens et al. (2013, 2015). When the system was at low risk of closure ($S < 0.1$), we assumed the system would be open and that lagoonal water level (η) would track mean seal level (MSL):

$$\eta = MSL$$

When the system was at high risk of closure ($S > 0.1$), we assumed the system would be closed and that lagoonal water level (η) would be largely determined by the starting conditions of the estuary mouth and net fluvial inputs (Q_{net}):

$$\eta_{t+1} = \eta_t + Q_{net}/Area_\eta;$$

$$Q_{net} = Q_{river} - Q_{evap};$$

Where η_{t+1} is the future lagoonal water level, η is current water level, Q_{net} is the sum of fluvial inputs and evaporation, and $Area_\eta$ is the surface area of the system at a given water level determined by system hypsometry.

For each of the 36 systems where we were able to apply our simple model for mouth dynamics, we calculated the percent of time that the system was expected to have high closure risk, and the hypothetical changes to water levels when we assume the system is closed. We aggregated this data by archetype in order to extrapolate to other systems without sufficient data to be included in this mouth dynamics analysis. To deal with the uncertainty of our mouth dynamics outputs, we added the predicted increases in the percentage of time that an archetype was at high risk of closure with 2050 and 2100 SLR and current estimates of percentage of the time these systems were closed (data provided by SCCWRP). The output provided an estimate of closure risk for 2050 and 2100, which were binned into the following categories: predominantly open (<40%); intermittently open/closed (>40%, <60%); and predominantly closed (>60%). Our mouth dynamics modeling outputs of hypothetical water levels were applied to these binned classes differently: predominantly open systems received no additional changes in water level resulting from mouth dynamics; intermittently open/closed systems received a dampened (0.5x) increase in water level; and predominantly closed systems received the full (1x) increase in water level associated with closed mouth state.

Marsh Hypsometry

Hypsometric curves were developed for all 104 systems using a digital elevation model (DEM) obtained from the 2009-2011 NOAA-CA Coastal Conservancy Coastal Lidar Project. This DEM is provided in raster format with a spatial resolution of 1 m^2. All spatial datasets were analyzed in ArcMap 10.3 (ESRI, Redlands, CA). The DEM raster was masked to estuary footprints, which were provided by SCCWRP. Once the DEM was clipped to the extent of each individual estuary, we converted elevation (z) to elevation capital (Z*):

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

Z* is the relative elevation within the tidal range and is a dimensionless ratio of elevation referenced to mean sea level (MSL) and mean high high water (MHHW) (Swanson et al. 2014). Z* was used in order to standardize estimates of elevation changes across estuaries with varying tidal datums, elevations and tidal ranges. Conversion of elevation to Z* was conducted using the *raster calculator* tool in ArcMap 10.3, using MSL and MHHW records from the nearest NOAA tidal station for each estuary (Table 3 & 4).

Using the Z* rasters for each system, we created hypsometric curves by calculating the frequency of raster cells ($1m^2$) that fall within certain elevational ranges. For the purpose of our assessment we used Z* bins of 0.05. Raster cell counts were conducted using the *hist* function in the R Package *raster* v2.5-2. Counts within each Z* bin were converted to area (km^2), which was cumulatively summed to provide the cumulative area (km^2) needed to create standard hypsometric curves (e.g., Figure 3).

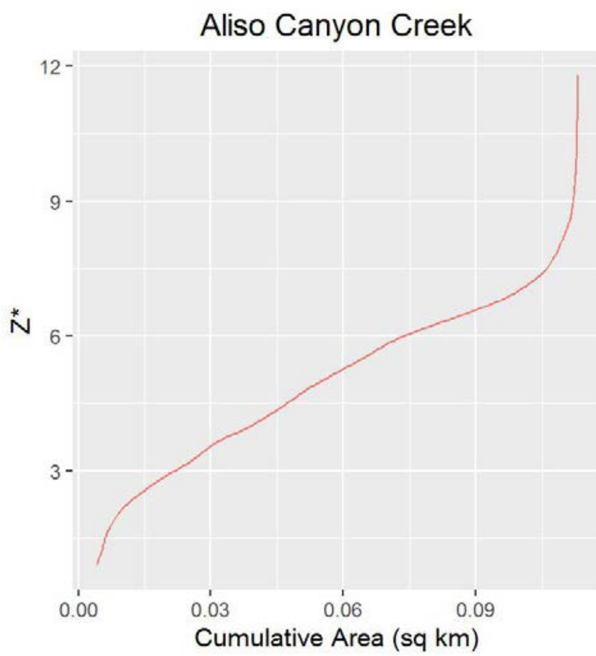

Figure 3. Hypsometric curve developed for Aliso Canyon Creek using the NOAA-CC LiDAR DEM.

Table 3. *NOAA station tidal data used in Z* calculations.*

Tidal Station	Station ID	Tidal Data (m NAVD88)					
		MLLW (m)	MLW (m)	MSL (m)	MHW (m)	MHHW (m)	HAT (m)
Santa Barbara, CA	9411340	-0.039	0.260	0.811	1.376	1.606	2.162
Gaviota State Park, CA	9411399	-0.028	0.269	0.809	1.359	1.583	2.149
Rincon Island, CA	9411270	-0.030	0.271	0.831	1.404	1.634	
Santa Monica, CA	9410840	-0.057	0.226	0.792	1.371	1.596	2.158
Cabrillo Beach, CA	9410650	-0.075	0.211	0.785	1.371	1.596	
Newport Beach, CA	9410580	-0.055	0.224	0.790	1.369	1.594	2.132
La Jolla, CA	9410230	-0.058	0.218	0.774	1.344	1.566	2.119
San Diego, CA	9410170	-0.132	0.154	0.765	1.388	1.613	2.222
Imperial Beach, CA	9410120	-0.074	0.200	0.765	1.340	1.563	

Table 4. Z calculations for each NOAA tidal station.*

Tidal Station	Station ID	Z* Range Upper Limit					
		Subtidal (z = MLLW)	Mudflat (z = MSL)	Low Marsh (z = MHW)	Mid Marsh (z = MHHW)	High Marsh (z = HAT)	Transition (z = HAT+)
Santa Barbara, CA	9411340	-1.069	0.000	0.711	1.000	1.699	2.083
Gaviota State Park, CA	9411399	-1.081	0.000	0.711	1.000	1.731	2.125
Rincon Island, CA	9411270	-1.072	0.000	0.714	1.000	1.699*	2.079
Santa Monica, CA	9410840	-1.056	0.000	0.720	1.000	1.699	2.078
Cabrillo Beach, CA	9410650	-1.060	0.000	0.723	1.000	1.669*	2.045
Newport Beach, CA	9410580	-1.051	0.000	0.720	1.000	1.669	2.048
La Jolla, CA	9410230	-1.051	0.000	0.720	1.000	1.698	2.083
San Diego, CA	9410170	-1.058	0.000	0.735	1.000	1.718	2.078
Imperial Beach, CA	9410120	-1.051	0.000	0.721	1.000	1.669*	2.051

Z* values shown here are determined using the equation presented above where Z* = (z -MSL)/(MHHW -MSL). Elevation (z) used in these calculations is expressed under each habitat type (z = ---), and can be found in Table 3. *Highest Astronomical Tide (HAT) not available; value assumed.

Changes in Elevation and Water Level

Model components for rSLR, accretion and mouth dynamics provide the key inputs needed for intermediate model calculations of changes in elevation and water level. We used 2016 (t_0) as our baseline for comparison to future SLR response for 2050 and 2100. Change in marsh elevation (ΔE) is determined by rSLR and accretion:

$$\Delta E_t = SL_t - A_t$$

Where SL_t is the change in sea level for a given time point ($t_{1,2}$=2050, 2100) and A_t is the total accretion by that time. SL_t and A_t are calculated from the rates described above by multiplying by the desired time period ($t_{1,2}$ - t_0) and converting to meters.

Change in water level ($\Delta\eta$) was determined by rSLR and mouth dynamics:

$$\Delta\eta_t = SL_t + \eta_{closed}$$

Where η_{closed} is the hypothetical change in lagoonal water level when a system is assumed to be closed (see section *Mouth Dynamics*). ΔE_t and $\Delta\eta_t$ were estimated for all 104 systems for both 2050 and 2100. These calculations are used in combination with the hypsometric curves to estimate habitat change.

Habitat Change

We used the intermediate model calculations for changes in elevation and water levels for 2050 and 2100, along with the hypsometric curves developed for each system to estimate areal changes in habitat arising from SLR. Our conceptual diagram indicates how ΔE and Δη were applied to hypsometric curves (Figure 3). Changes in elevation act upon the hypsometric curve itself, essentially increasing the Z* values by multiplying by ΔE (which has been converted from meters to Z* using the local tide datum). This manipulation of the hypsometric curve does not alter the total area of the marsh, but rather "raises" the marsh elevation capital over time.

Changes in water level are not directly applied to the hypsometry but instead are used to manipulate the Z* ranges that correspond to different marsh habitats (Figure 3, Table 5).

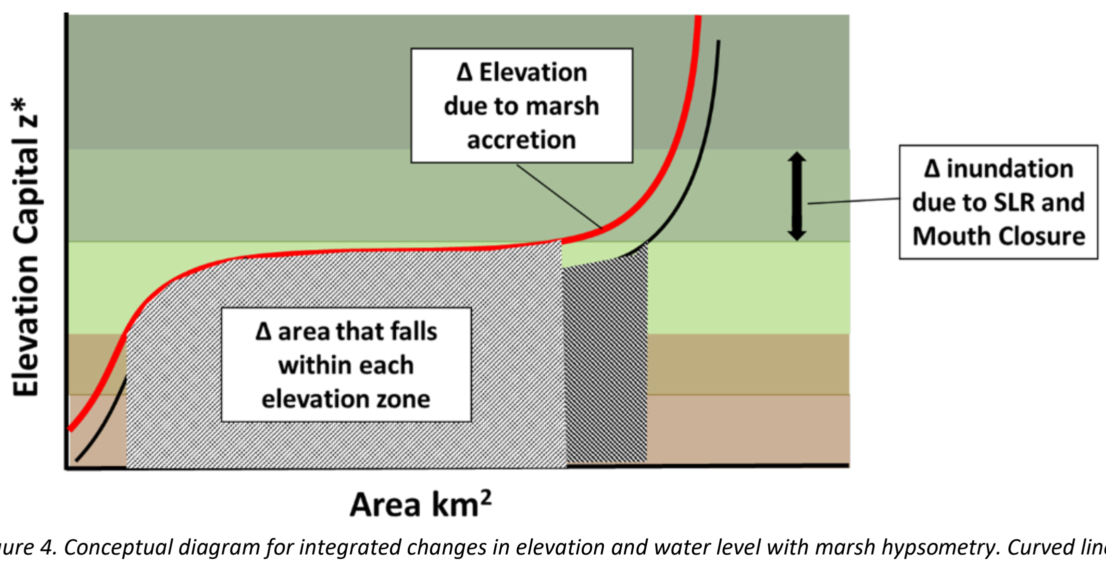

Figure 4. Conceptual diagram for integrated changes in elevation and water level with marsh hypsometry. Curved lines represent current (black) and future (red) marsh hypsometry which have been altered by changes in elevation. Background colors (brown – green) represent different marsh zones. Double-ended arrow indicates the changes in the marsh zone limits which arise with changes in water levels. Shaded areas represent that change in area that fall within a given marsh zone for current (left crosshatch) and future (right crosshatch) scenarios.

Table 5. Z range upper limits for each marsh zone under current, 2050 and 2100 conditions. Changes to z* are determined by site-specific changes in water level estimated in our analysis. Example values shown for the Aliso Canyon estuary in San Diego county. Z* for this site is based on the Newport Beach, CA NOAA tidal station.*

	Z* Range Upper Limit		
Marsh Zone	**Current**	**2050**	**2100**
Subtidal	-1.05	-0.38	1.16
Intertidal Mudflat	0.00	.66	2.19
Low Marsh	0.72	1.38	2.91
Mid Marsh	1.00	1.66	3.19
High Marsh	1.67	2.33	3.86
Transition	2.6	3.33	4.86

We calculated the area within each zone (subtidal, intertidal mudflat, low marsh, mid marsh, high marsh and transitional) for each of the 104 estuaries for current conditions, as well as 2050 and 2100 SLR scenarios. Area calculations were performed using the *sum* function in the R Package *base* v3.2.3.

Habitat Change Index

We developed an index to score the relative effect of SLR on each of the 104 systems. The Habitat Change Index (HCI) represents the total expected absolute percent change in all habitat types (Δ%) within a wetland. We then normalize the total percent change by the maximum possible percent change (200%) to allow comparison across wetlands (Equation 1). HCI scores were grouped into low, medium) and high qualitative bins to reflect varying levels of habitat change. Because HCI is an index of overall change, increases in HCI for the "Wetland Migration/All Areas" scenario reflect potential gains in vegetated marsh and flat habitats and potential losses in subtidal habitats.

$$HCI = \left(\frac{\sum|\Delta\%|}{200\%}\right) * 10$$

Effect of Active Management Beyond Facilitated Expansion

In addition to the three scenarios of wetland expansion, we predicted how SLR-induced habitat change may be mitigated by simulating four additional management actions in the model:

- <u>Increasing accretion rates</u> to 12.2 (± 5) mm yr^{-1} and 16.6 (± 5) mm yr^{-1} to keep pace with SLR
- <u>Implementing one-time thin-layer sediment augmentation</u> at a depth of 23.4 (± 10) cm following the work of Thorne et al. (2016a) at the Seal Beach National Wildlife Refuge.
- <u>Managing estuary mouth dynamics</u> by either allowing systems to close or actively maintaining systems to be open.
- <u>Reconnecting fragmented systems</u> that were historically a single wetland complex.

Uncertainty Analysis

We conducted an uncertainty analysis to address three areas of uncertainty in our model: 1) Absence of information; 2) Errors of measurement; and 3) Sensitivity to model parameters. Specific sources of uncertainty include SLR projections, accretion data availability, mouth dynamics assumptions, vertical datum accuracy, and the archetype-based extrapolation of data throughout the region. To address uneven data availability, we quantified data confidence for each of the key model inputs for each of the 104 sites in the SCB (Table 6). These scores reflect the origin of input data and were assigned for SLR, accretion and mouth dynamics model components. As part of the uncertainty analysis, we propagated errors (e.g., standard error, standard deviation, 95% confidence intervals) associated with each of the model inputs through the model to determine the potential errors in the habitat change output. This error analysis provides us with a bookended range of potential habitat change. Lastly, our sensitivity analysis allows us to identify the importance of each model input in determining the habitat change output. To do this, we modified each input by ±50% while leaving all other inputs unchanged and propagating the change in input through the model. This sensitivity analysis provides us with an estimate of the % change in habitat caused by modifying the inputs by ±50%.

Table 6. Uncertainty score rationale.

Uncertainty Score	Description	Example
1	Site-specific data of high quality is available	Measured accretion rate 7.6 mm yr^{-1} at Tijuana Estuary
2	Data is limited; regional data derived from site-specific measurements/observations were extrapolated using the archetype framework	Regional literature review produced accretion rates by archetype which were used for data-poor sites
3	Regional defaults	NRC 2012 SLR projections

Statistical Analyses

We examined statistical significance of differences in accretion estimates for various groupings, i.e., site, marsh zone, archetype, using one-way analysis of variance (ANOVA) for complete random designs (*aov* in R Package *stats* v3.2.3). To test multiple groupings, i.e., marsh zones within sites, we used one-way ANOVA for a randomized block design. Assumptions of normality and homogeneity of variance were tested using the Shapiro-Wilk W test (*shapiro.test* in R Package *stats* v3.2.3) and the Bartlett's test (*Bartlett.test* in R Package *stats* v3.2.3), respectively. Failure to meet the assumptions resulted in data being log-transformed. Failure to meet assumptions following transformation resulted in the use of the Kruskal-Wallis Rank Sum test for nonparametric data (*kruskal.tes*t in R Package *stats* v3.2.3). Significant results ($\alpha = 0.05$) prompted a post hoc analysis using Tukey's Honest Significant Difference test (*TukeyHSD* in R Package *stats* v3.2.3), aided by visualization of significance groupings (*HSD.test* in R Package *agricolae*).

Results

Accretion

Our literature review produced 110 records of accretion from 15 sources. Estimates of accretion were found for 9 of the 104 systems of interest in this study (Figure 4). Accretion was found to be significantly different between sites (p=0.015). Accretion estimates were not significantly different when grouped by marsh zone (p=0.174) (Table 5). When considering the interaction of site and marsh zone, accretion was significantly impacted by site (p=0.016), as well as zone (p=0.021). Table 7 illustrates the data gaps in assessing marsh accretion rate by zone for this region.

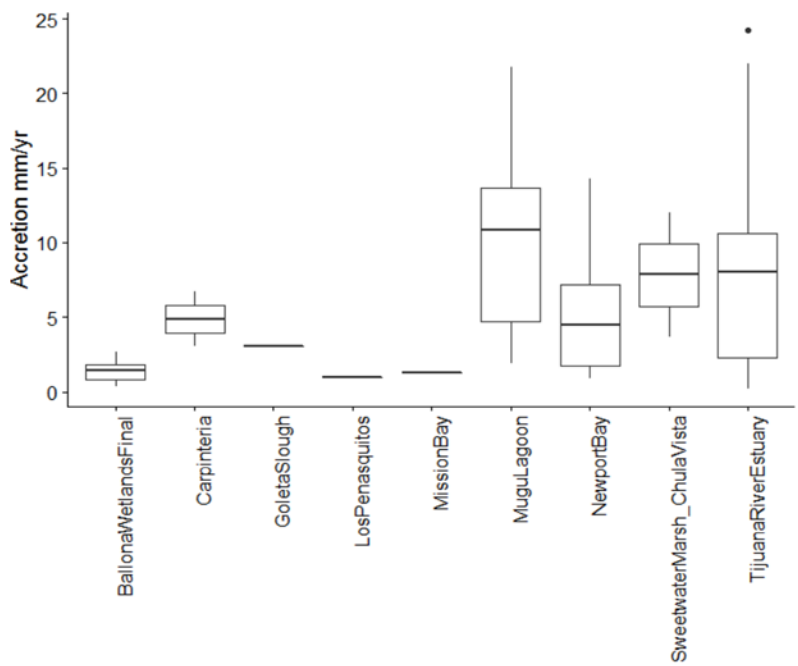

Figure 5. Boxplot of accretion estimates for a subset of estuaries in Southern California.

Table 7. Accretion estimates by marsh zone.

Marsh Zone	Accretion ± SD (mm yr^{-1})	n
Subtidal	1.75	1
Low	3.16 ± 1.44	4
Mid	4.77 ± 4.76	6
High	14.46 ± 10.38	2
Unspecified	3.23 ± 1.65	5

Table 8. Accretion estimates summarized by archetype.

Archetype	Zone	Accretion ± SD (mm yr^{-1}) By Zone	Accretion ± SD (mm yr^{-1}) Total	n
3	Low	3.63	3.63	1
4	Mid	12	9.55 ± 9.40	4
	High	21.8		
5	Low	4.8	4.68 ± 2.94	5
	Mid	4.83 ± 5.49		
6	Low	1.38	1.2 ± 0.65	3
	Mid	0.49		
7	Low	2.83	3.84 ± 2.32	5
	Mid	3.24 ± 2.82		
	High	7.13		

Mouth Dynamics

From our mouth dynamics modeling analysis, we found that certain systems are more susceptible to increases in the likelihood of closure with increases in sea level (Table 9). For example, the small creek and small lagoon archetypes are most at risk for increased likelihood of closure by 2050 and 2100, while larger systems with substantial fluvial inputs are more likely to remain open. When systems were expected to close, we determined the associated changes to water levels for each archetype (Table 9).

Table 9. Increased likelihood of high closure risk and the resultant increases in water levels when a system is presumed to be closed.

Archetype		2050 Δ Likelihood of Closure (%)	2050 Δ Lagoonal water level (m)	2100 Δ Likelihood of Closure (%)	2100 Δ Lagoonal water level (m)
Small Creek	1	+13%	0.43	+27%	1.38
Small Lagoon	2	+8%	0.43	+48%	1.55
Intermittently Open Estuary	3	+3%	0.42	+14%	1.41
Large Perennially-Open Lagoon	4	+7%	0.42	+21%	1.38
Large River Valley Estuary	5	0%	0	0%	0
Fragmented River Valley Estuary	6	No Data	No Data	No Data	No Data
Open Bay/Harbor	7	No Data	No Data	No Data	No Data

Changes calculated using 2016 as the baseline. Values for change in lagoonal water levels represent the contribution of mouth dynamics alone; these values will be combined with inundation from SLR in order to estimate total increases in water level in the estuary.

Habitat Change

Absent intervention, sea level rise will result in substantial loss of coastal wetlands

Absent any intervention, 407 ha of vegetated marsh and unvegetated flats will be lost with 0.6 m of SLR (by 2050 under current projections) and 1,580 ha will be lost with 1.6 m of SLR (by 2100 under current projections) (Figure 6). The predicted losses represent 12% and 48% of existing vegetated marsh and unvegetated flat areas, which is currently estimated to be 70% of wetland area in the region. The predicted habitat distribution for 0.6 m and 1.6 m of sea level rise are based on the current existing footprint without marsh migration or any other management intervention. Under this scenario, subtidal areas will increase by 488 ha and 1,870 ha with 0.6 m and 1.6 m of SLR, respectively.

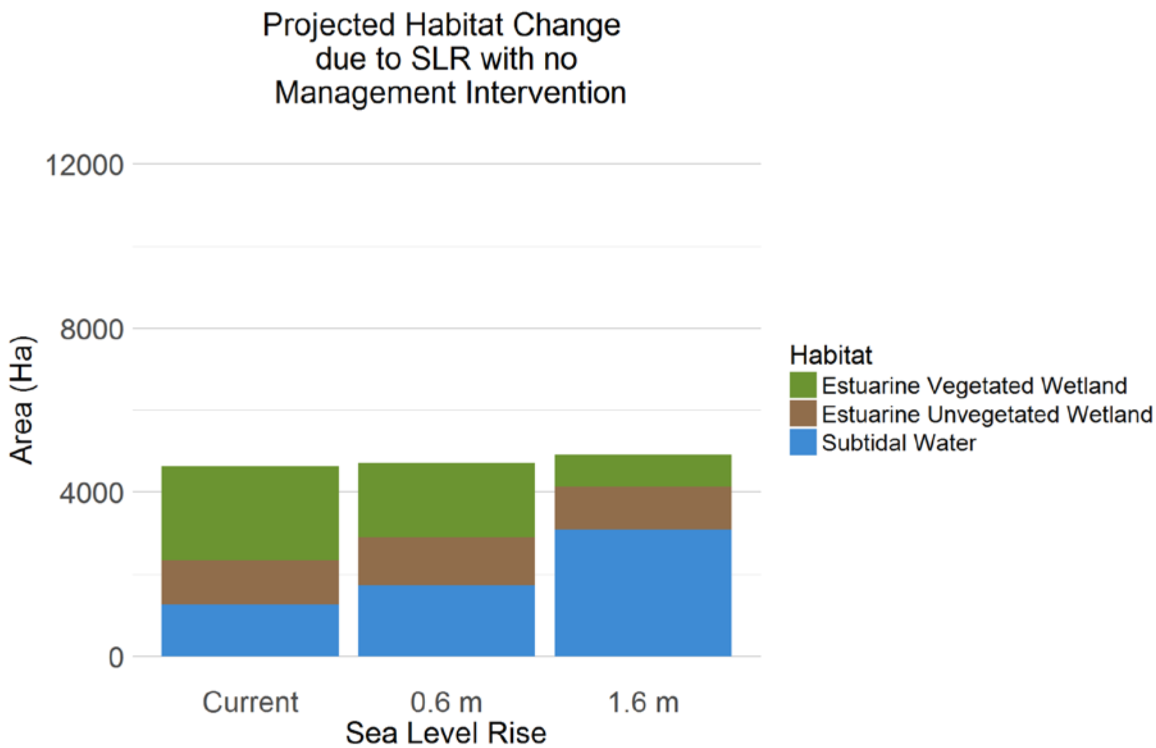

Figure 6. Habitat distributions under current SLR scenarios illustrate change in habitat distribution with SLR and no wetland expansion.

Opportunities exist today to restore and expand wetlands

One way to prevent predicted losses of wetlands to sea level rise is to restore and expand the wetlands that we have today. There are existing opportunities to increase current wetland extent by allowing wetlands to expand into adjacent areas with suitable elevations. Allowing expansion into undeveloped areas could increase the current coverage of vegetated marsh and unvegetated flats by an additional 2,900 ha, an 85% percent increase from existing wetland area. Allowing wetland expansion into any area with suitable elevation, regardless of land use, would increase vegetated marsh and unvegetated flat extent by an additional 5,450 ha, an increase of 160% from existing wetland area (Figure 7). It is important to note that expansion of current wetland habitats would require removal of physical barriers such as roads, levees and other infrastructure.

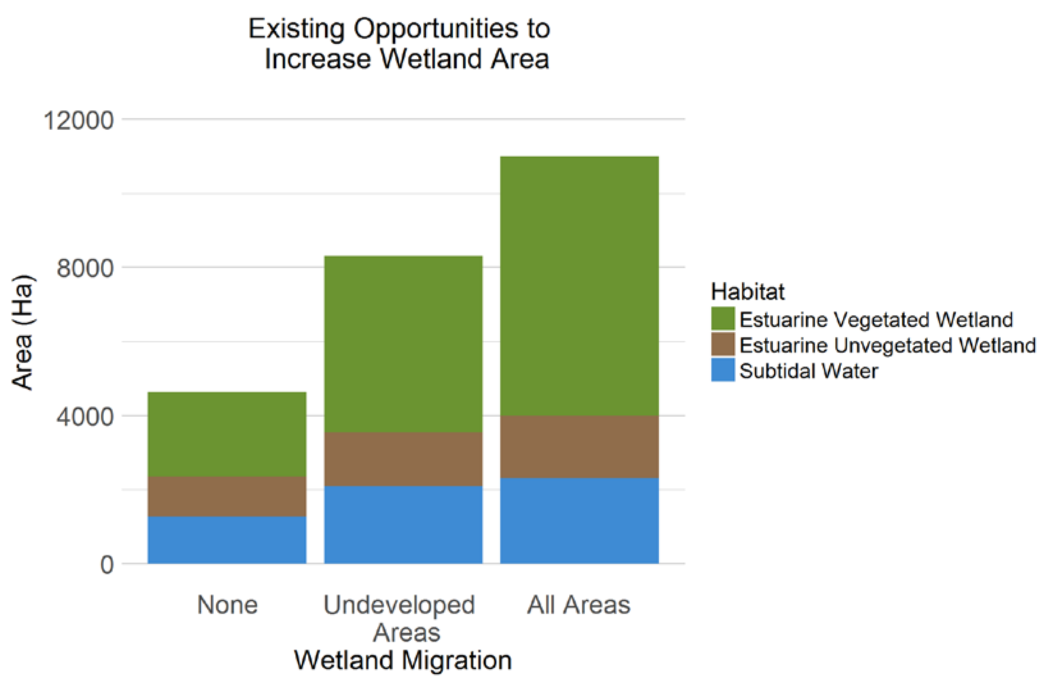

Figure 7. Current opportunities for wetland habitat expansion. Bars illustrate opportunities for wetland gain by allowing wetland footprints to expand to suitable elevations under current sea levels. "None" corresponds to the "No Wetland Migration/Existing Footprint" scenario; "Undeveloped Areas" corresponds to the "Wetland Migration/Avoid Developed Areas" scenario; "All Areas" corresponds to the "Wetland Migration/All Areas" scenario.

Maintaining wetlands in the future depends on wetland expansion

Beyond expanding our current wetlands to increase their resiliency to sea level rise, allowing wetlands to migrate with rising sea levels will offset expected habitat losses. According to the model, areas that are currently wetlands will be converted to subtidal habitat. However, areas that are currently uplands can become wetlands if we facilitate their migration.

Absent any intervention, 407 ha (12% of existing area) of vegetated marsh and unvegetated flats will be lost with 0.6 m of SLR (by 2050 under current projections) and 1,580 ha (46% of total area) will be lost with 1.6 m of SLR (by 2100 under current projections) (Figure 6). The predicted habitat distribution for 0.6m and 1.6m of sea level rise are based on the current existing footprint without marsh migration or any other management intervention.

With 0.6 m of sea level rise, 3,060 ha of current upland habitat could become vegetated marsh and flats if wetlands were able to expand into currently undeveloped areas (wetland migration/avoid developed areas) (Figure 8). If wetlands were able to expand into all areas with appropriate elevations (wetland migration/all areas), 6,240 ha of upland area could be available for wetland habitat. Wetland migration will require active intervention and restoration actions to remove barriers and facilitate wetland growth in newly accessible areas.

With 1.6 m of SLR, 3,587 ha of current upland habitat could become vegetated marsh and flats if wetlands were able to expand into currently undeveloped areas (wetland migration/avoid developed areas). If wetlands were able to expand into all areas with appropriate elevations (wetland migration/all areas), 7,869 ha of upland area could be available for wetland habitat.

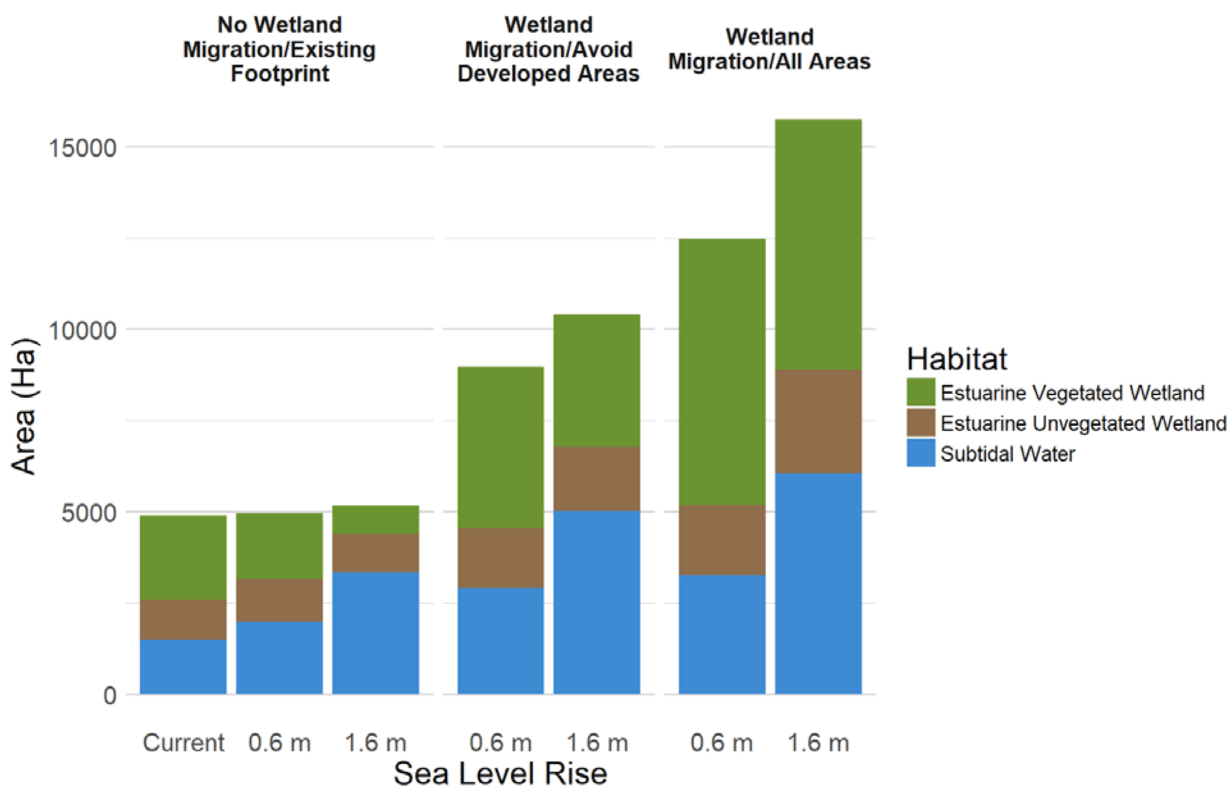

Figure 8. Effect of SLR varies based on allowable wetland expansion. Each panel shows habitat distribution under current, 0.6 m SLR (2050), and 1.6 m SLR (2100) based on the existing footprint, expansion but avoiding developed areas, and expansion into all areas.

Changes in habitat resulting from sea level rise will vary by wetland archetype and subregion with the greatest changes expected for large river valleys and the southern portion of the region.

Sea level rise will affect wetland archetypes differently based on their size, current habitat composition (Figure 9). Furthermore, differences in available upland transition space cause varying opportunities for wetland migration. Size and habitat composition differences make it difficult to compare SLR effects across different archetypes. To facilitate comparison of relative effects of SLR across systems and archetypes, we created a Habitat Change Index (HCI) (Equation 1) to represent the relative percent change in habitat composition within a wetland (Δ%).

$$HCI = \left(\frac{\sum|\Delta\%|}{200\%}\right) * 10$$

To create a standardized index of change, we normalized the percent change by the maximum possible percent change (200%), to allow comparison across wetlands. The percent of habitat change is calculated by subtracting the predicted future habitat composition from the current habitat composition. A high HCI score represents a wetland with a large change in habitat composition. Some changes in habitat composition were in the positive direction, meaning that percent wetland coverage would increase with sea level rise, but for the most part changes in habitat composition were in the negative direction, meaning that there were losses in wetland coverage. HCI scores were grouped into low, medium and high qualitative bins to reflect varying levels of habitat change.

The HCI demonstrates that there are differences in wetland habitat change within archetypes and by region (Figures 10 and 11). Most archetypes exhibit modest effects under 0.6 m SLR, and more pronounced effects under 1.6 m SLR. The exception is small creek mouths and small lagoons, which show much greater response under lower levels of SLR. Small lagoon systems often lack appreciable transition zones due to steep topography surrounding the wetlands. In these small lagoons, we predict 100% of the wetland area will be converted to subtidal habitats with 1.6 m of SLR. Similarly, small creeks may experience an increase in both subtidal habitat and vegetated marsh and unvegetated flats, but only when upland transition zones are made available. Larger systems with a relatively low proportion of subtidal habitat (e.g. large river valleys and fragmented river valleys) have the highest HCI values under 1.6 m SLR (but modes change under 0.6 m SLR). However, these larger archetypes, including intermittently opening/closing estuaries, perennially open lagoons and river valley estuaries, could be able to maintain current wetland coverage, or experience an increase in wetland habitats, if migration into adjacent transition zones is made available. The net maintenance or gain of wetland habitats in the larger wetlands will be determined by low elevation topography of neighboring areas and the availability of migration, as well as relatively high rates of sediment accretion. Spatially, the largest HCI values (i.e. most change) is predicted for wetlands in the San Diego subregion; this difference is largely a function in the concentration of larger systems in this region relative to the more northerly subregions.

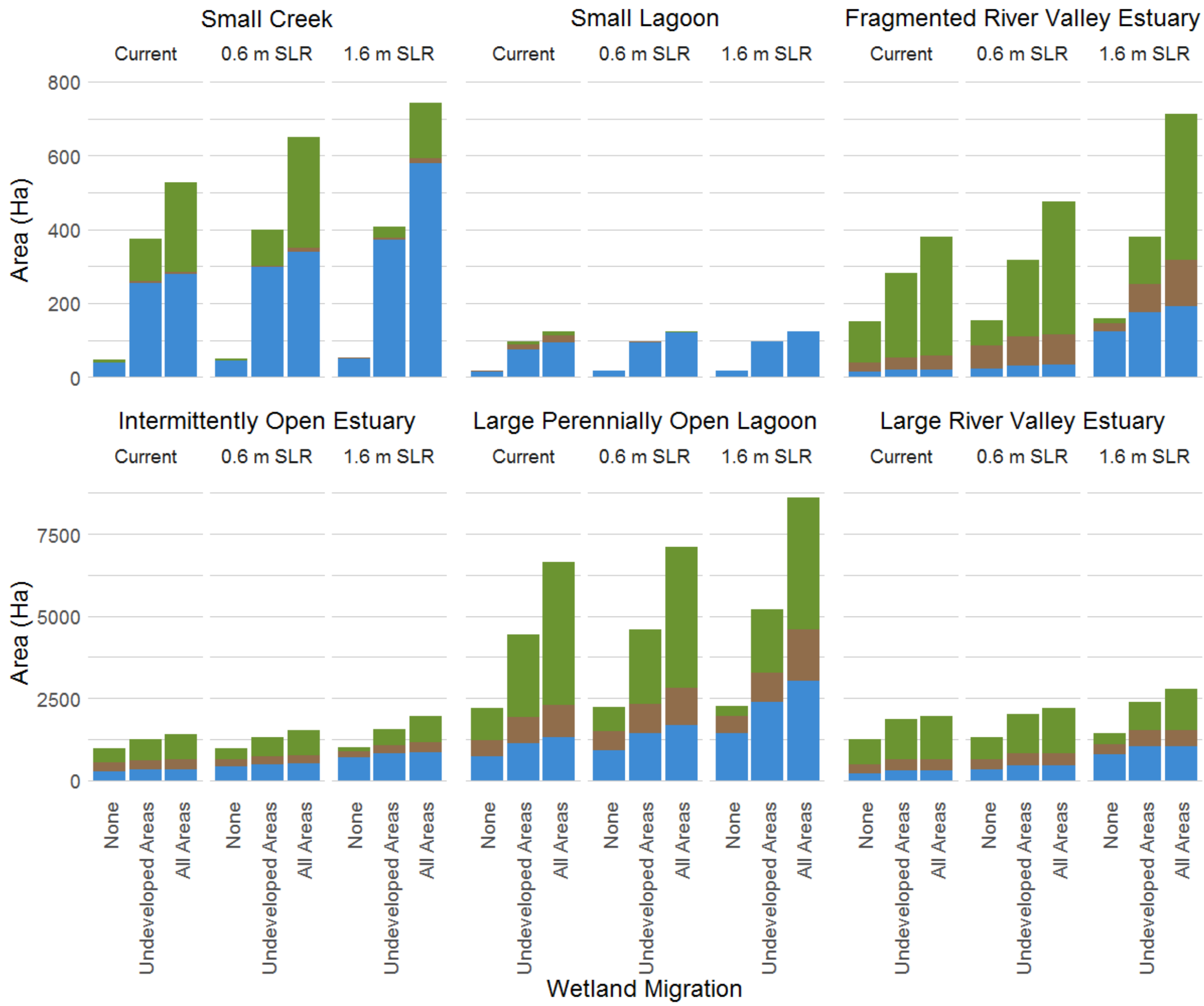

Figure 9. Predicted habitat change under 0.6 m (2050) and 1.6 m (2100) SLR for each archetype. Effects are shown based on current footprint, expansion and avoiding developed areas and expansion into all areas. "None" corresponds to the "No Wetland Migration/Existing Footprint" scenario; "Undeveloped Areas" corresponds to the "Wetland Migration/Avoid Developed Areas" scenario; "All Areas" corresponds to the "Wetland Migration/All Areas" scenario.

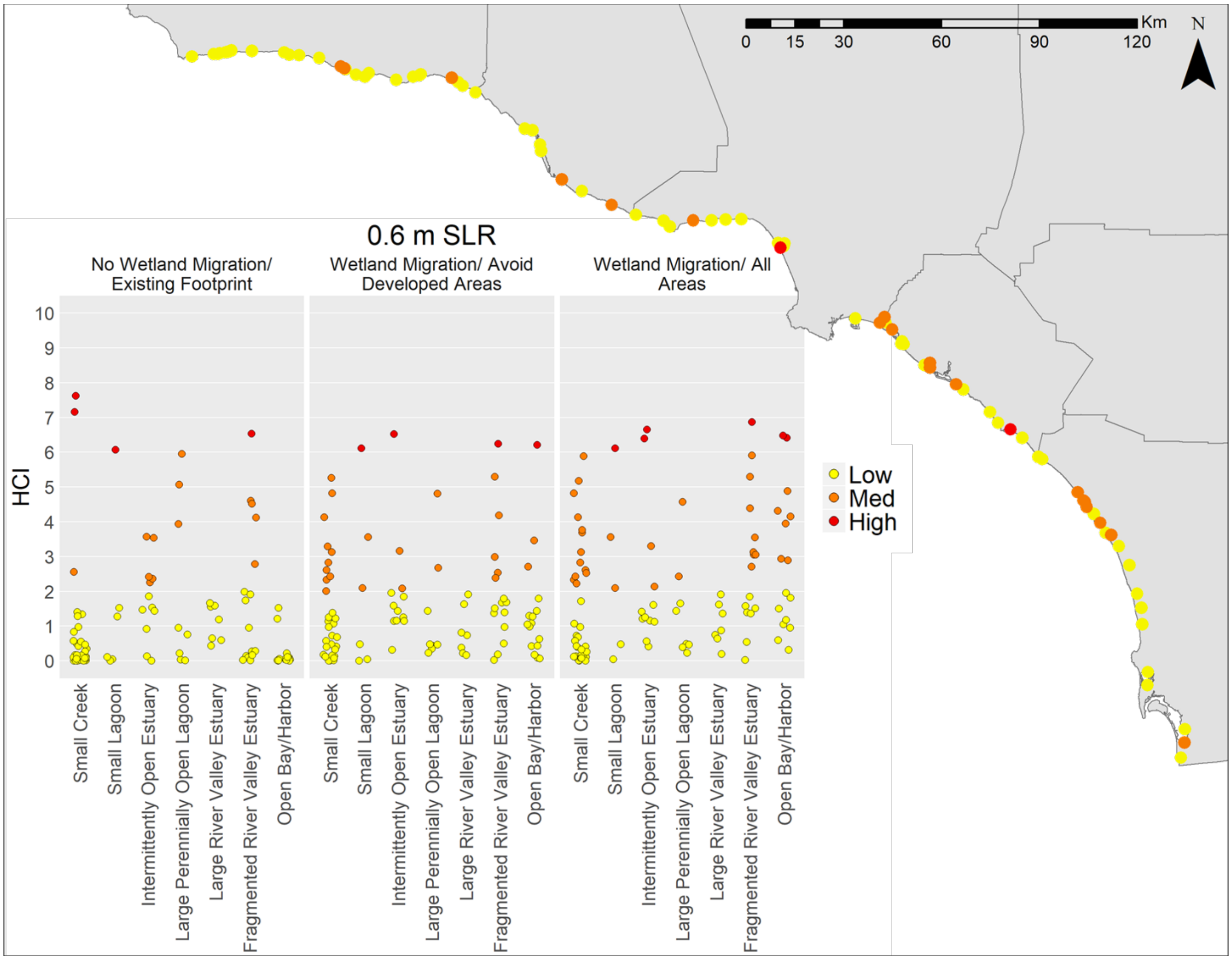

Figure 10. Habitat Change Index (HCI) scores for each archetype and each wetland footprint with 0.6 m of sea level rise. Map displays HCI scores for individual sites under the wetland migration/avoid developed areas scenario. Large open bays and harbors are not shown on the map for clarity.

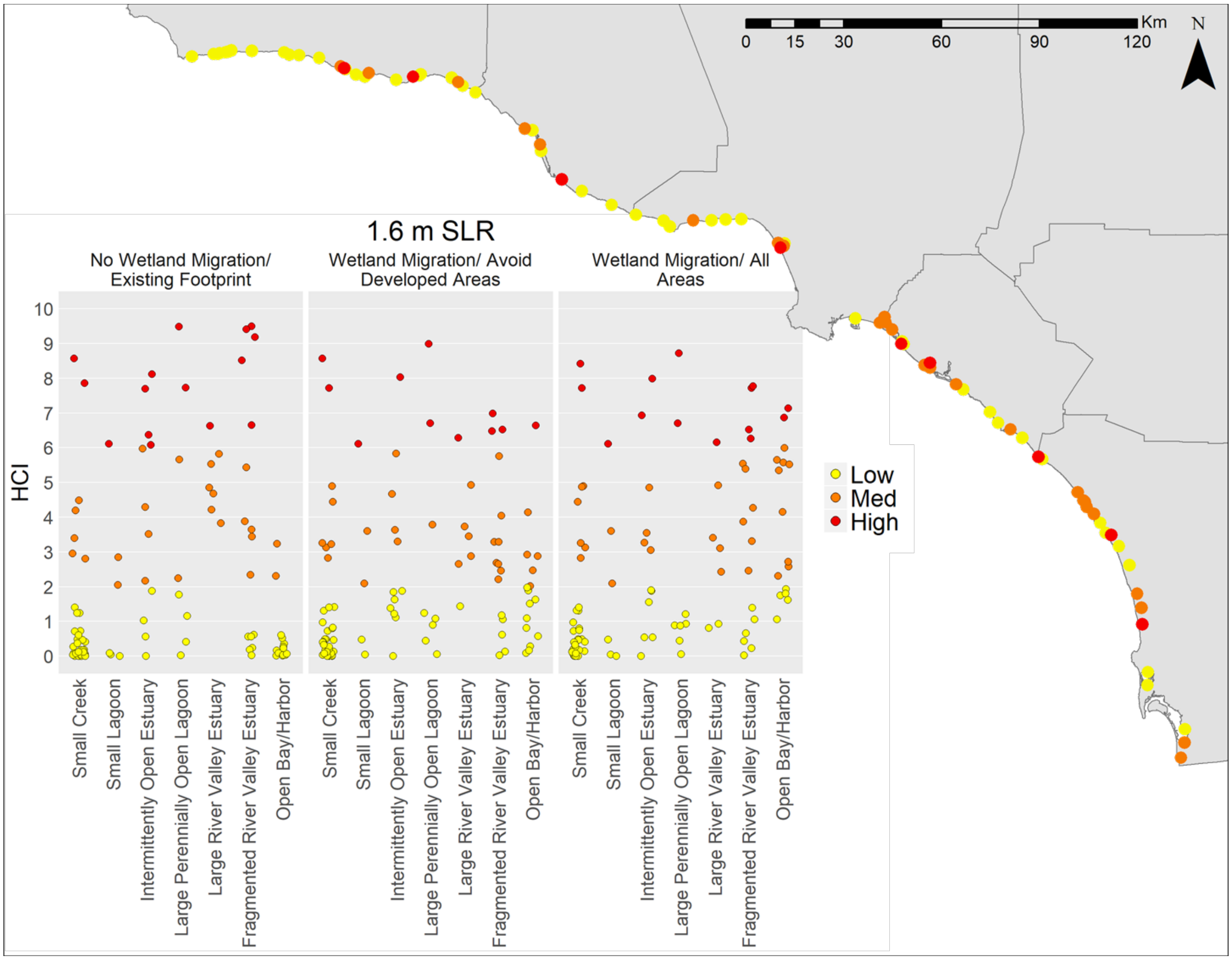

Figure 11. Habitat Change Index (HCI) scores for each archetype and each wetland footprint with 1.6 m of sea level rise. Map displays HCI scores for individual sites under the wetland migration/avoid developed areas scenario. Large open bays and harbors are not shown on the map for clarity.

Management actions will reduce the adverse effects of sea level rise

In addition to running the model with three wetland footprints in order to quantify the effects of wetland migration, the model also demonstrated that the following four management actions could reduce sea level rise-induced habitat losses.

<u>Increasing sediment accretion rates</u> to 12.2 (± 5) mm per year and 16.6 (± 5) mm per year to keep pace with sea level rise.

Increasing annual sediment accretion rates would have a moderate effect on habitat change. If wetlands were not able to migrate into adjacent areas (no wetland migration/existing wetland footprint), augmenting annual sediment accretion rates to match a sea level rise of 0.6 m (12.2 mm of sediment per year), could save an additional 9% of vegetated marsh and unvegetated flat (271 ha) over the entire region (Figure 12). If marshes were able to migrate into adjacent undeveloped areas (wetland migration/avoid developed areas) and sediment accretion rates were augmented to keep pace with sea level rise, an additional 12% of vegetated marshes and flats (367 ha) could be saved.

<u>Implementing one-time thin-layer sediment augmentation</u> at a depth of 23.4 (± 10) cm following the work of Thorne et al. (2016a) at the Seal Beach National Wildlife Refuge.

With 0.6m of sea level rise, a one-time sediment augmentation event is comparable to increasing the annual sediment accretion rate (Figure 12). However, with 1.6 m of sea level rise, increasing annual accretion rates will increase wetland habitat by 25% more than the one-time sediment deposition (Figure 13).

<u>Managing tidal mouth dynamics</u> by either allowing inlets to open and close naturally or actively maintaining inlets constantly open.

Managing tidal inlet dynamics will have variable impacts at the regional scale. Habitats may change by up to 40% at individual systems due to changes in mouth management. However, when combined across the region, expected gains and losses balance out and result in no overall effect.

<u>Reconnecting fragmented systems</u> that were historically a single wetland complex.

Reconnecting systems that were once historically whole but have become fragmented would provide an additional 18% of wetland habitat with 0.6 m of sea level rise and under a scenario where no wetland migration could occur. Simultaneously allowing systems to expand into adjacent undeveloped areas while also reconnecting fragmented systems would provide an additional 33% of wetland habitat with 0.6 m of sea level rise.

With 1.6 m of sea level rise, system reconnection in conjunction with allowing wetland expansion into undeveloped areas would provide an additional 27% of wetland habitat. Without allowing wetland migration, system reconnection alone would provide only an additional 1% of wetland habitat with 1.6 m of sea level rise.

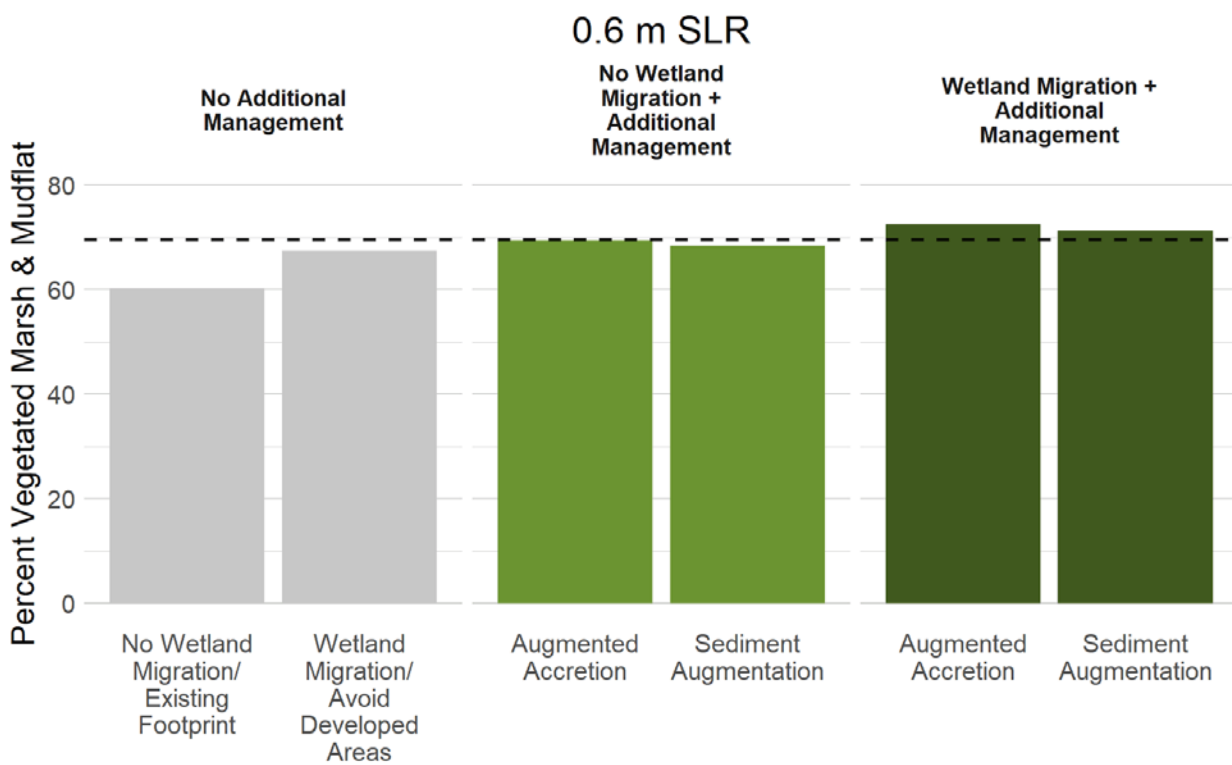

Figure 12. Effects of increased annual accretion and one-time sediment augmentation with 0.6 m of sea level rise. The dashed line represents the current baseline of 70% vegetated marsh and unvegetated flats. The figure shows three wetland footprints and two sediment augmentation scenarios.

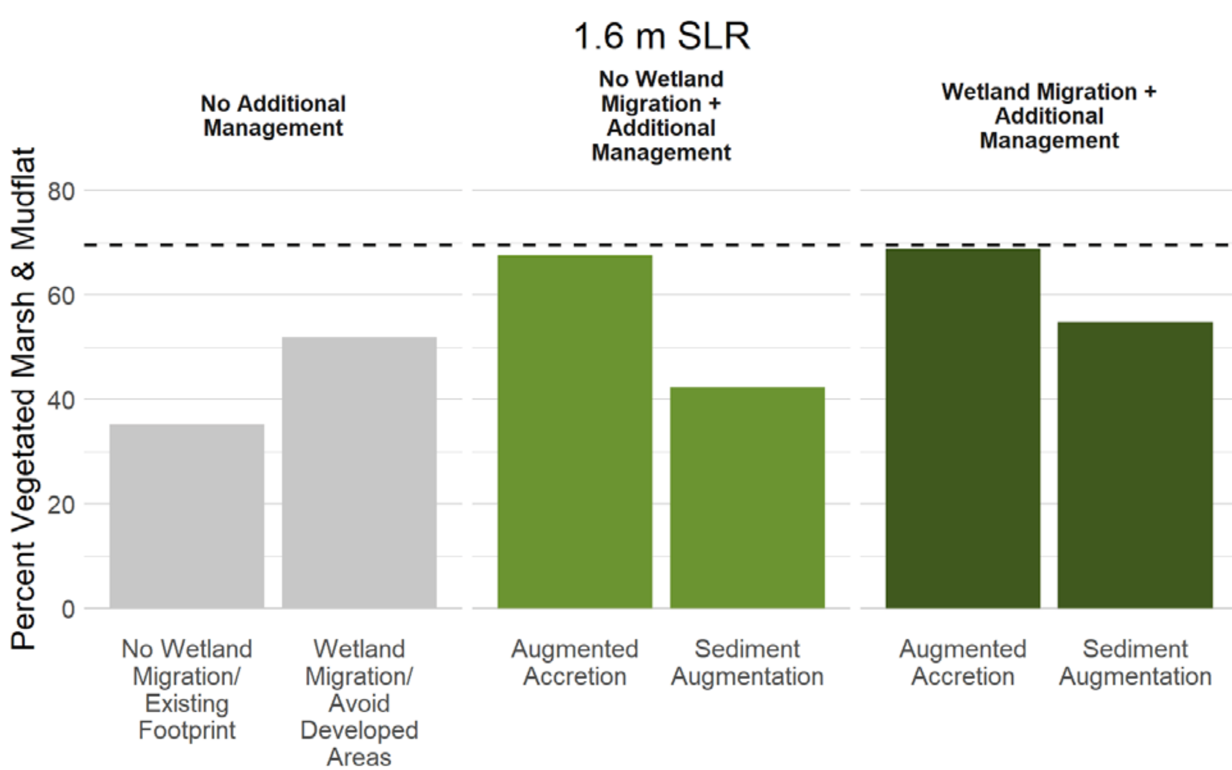

Figure 13. Effects of increased annual accretion and one-time sediment augmentation with 1.6 m of sea level rise. The dashed line represents the current baseline of 70% vegetated marsh and mudflats. The figure shows three wetland footprints and two sediment augmentation scenarios.

Uncertainty

We conducted an uncertainty analysis with three components: 1) Data Confidence; 2) Errors Analysis; and a 3) Sensitivity Analysis. We quantified data confidence for each of the 104 sites in the SCB. Sites with high data confidence were data-rich, while sites with low data confidence were lacking data. We found that data confidence was low for 42 sites, medium for 52 sites, and high for only 10 sites (Figure 14).

The error analysis revealed the ranges of potential habitat outputs when input error was propagated though the model (Figure 15). When the bookended range was compared to the original model outputs (black lines) for each habitat type for 2050 and 2100, we found that the predictions for subtidal areas are high within the range and that the predictions for vegetated areas are low within the range, while predictions for unvegetated mudflats are in the middle of the range. In all, predictions for 2100 exhibits higher uncertainty due to model inputs than those for 2050.

The sensitivity analysis revealed that model outputs are most sensitive to SLR, followed by accretion and mouth dynamic inputs (Figure 16). Increasing accretion inputs resulted in additional vegetated marsh areas, while decreasing accretion resulted in additional subtidal and unvegetated mudflat areas. Increasing the water levels associated with mouth closure resulted in increases of subtidal areas and decreases in vegetated/unvegetated marsh areas; conversely, decreasing this input results in increases in vegetated/unvegetated marsh areas. Increasing SLR inputs by 50% caused declines in vegetated/unvegetated marsh areas and increases in subtidal areas. Decreasing SLR inputs by 50% caused gains in vegetated marsh areas, but are not realistic expectations. Overall, the sensitivity of model outputs increases from 2050 to 2100

Figure 14. Data confidence scores reflecting data availability in the region.

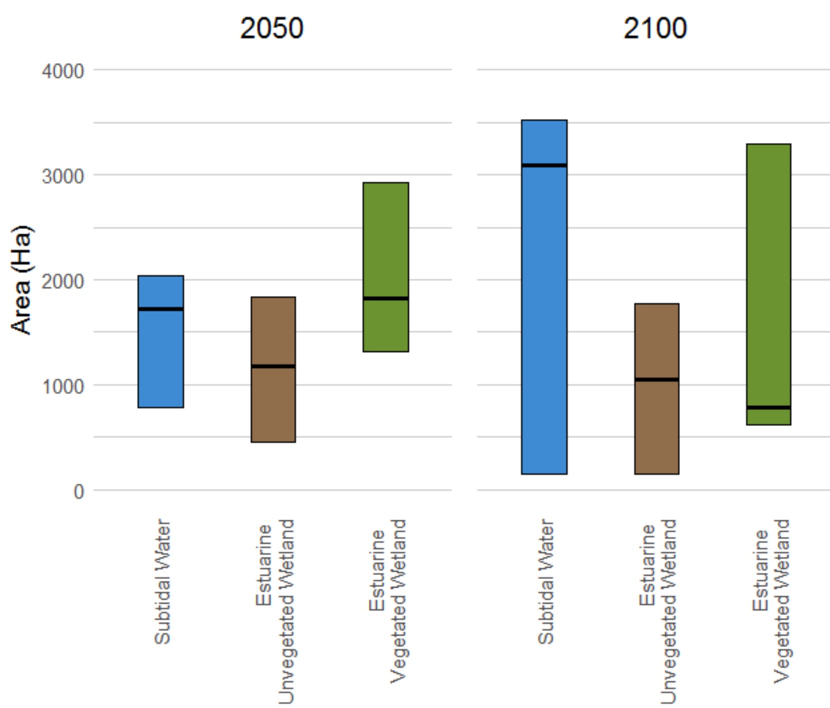

Figure 15. Bookended range of the potential area for each wetland habitat following our error analysis. Floating bars represent the minimum and maximum areas predicted when errors were propagated through the model. Black lines represent the area of each habitat originally predicted by the model.

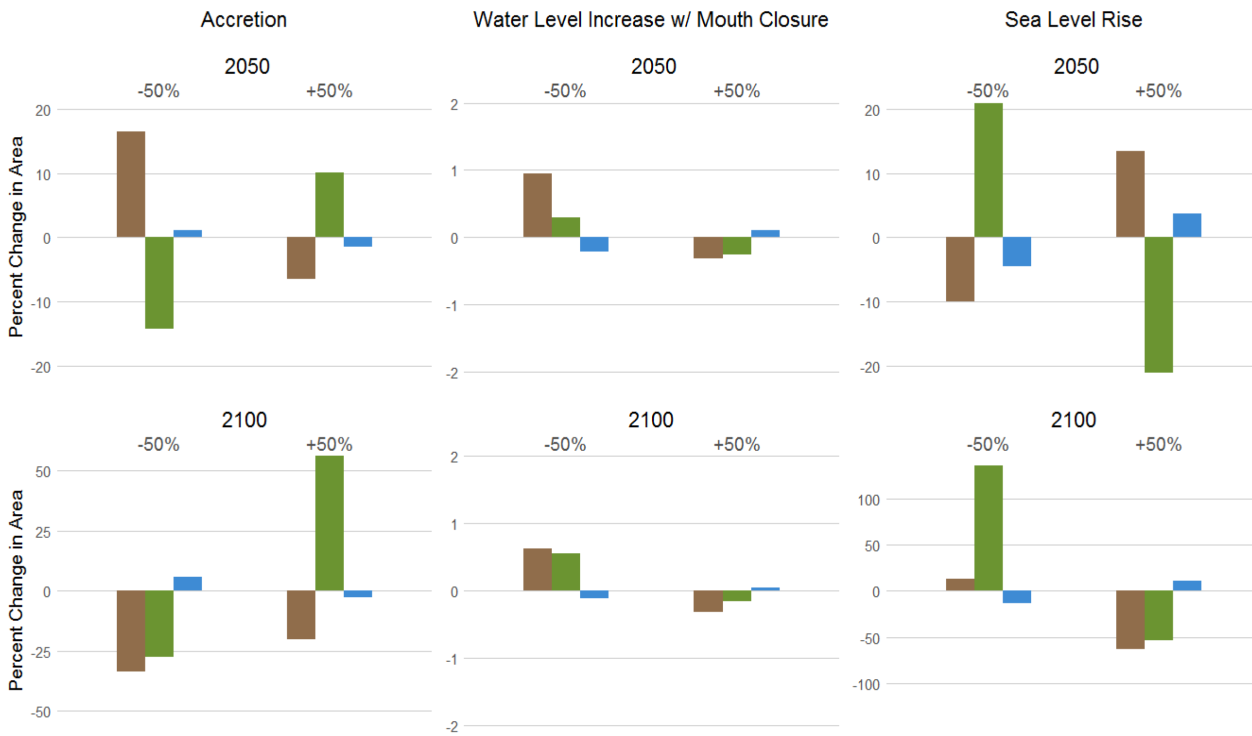

Figure 16. Percent change in habitat area caused by sensitivity to model inputs. Changes to subtidal (blue), unvegetated mudflat (brown), and vegetated marsh (green) habitats when model inputs (Accretion, Mouth Dynamics, SLR) are increased (+50%) or decreased (-50%). Note varying y-axis scales in plot panels.

Significance

Through this process, we produced a quantitative estimate of relative response to SLR, specific for the southern California region that is based on the dynamic processes that define local estuaries. Our modeling framework incorporates regionally-important factors that contribute to SLR response, such as mouth dynamics, which have been lacking in previous models (Swanson et al. 2014, Lentz et al. 2016). This approach may be relevant for assessing SLR response in estuaries in other Mediterranean settings across the globe, including South Africa and Australia, which are subject to similar processes (Jacobs et al. 2011). Based on use of archetypes, the model should provide a regionally applicable screening tool for application to a wide range of coastal systems, which may have differential responses to SLR (Lentz et al. 2016). The model also has the potential for increased parameterization, or the inclusion of site-specific data when it becomes available, which will decrease the uncertainty in model outputs and provide users with improved estimates of SLR response at a given site. Because this effort uses site estimates of habitat change to determine future resiliency, it strikes the balance between large-scale vulnerability assessments and site-specific marsh response models. Regional SLR response is often determined using data that is too coarse to capture the vulnerability of individual sites making it difficult to apply the results for local planning. More detailed modeling of marsh response is often conducted at larger, well-studied sites, and is too time and labor intensive to repeat for many systems within a region (e.g., Thorne et al. 2016). This means that many (smaller) estuaries are often under-represented in both local and regional planning efforts. Use of the archetype framework allows us to deal with these data gaps and leverage regional and site assessments. In doing so we provide increased resolution of SLR vulnerability, at scales relevant to regional and local management. We anticipate that the regional SLR response model will provide screening level assessment that can be used to prioritize and support more detailed site-specific investigations, and will provide a platform to prioritize future work based on greatest needs or uncertainties. Finally, through this effort we were able to compile regional data sets that were not readily available. These data sets will serve as a resource for local managers to support their planning and decision making.

Outreach

This work will contribute valuable information to the large body of stakeholders who research, manage, and regulate southern California coastal wetlands within the context of climate change. The work presented here is part of the ongoing SCWRP RSU effort to ensure the future resiliency of coastal system throughout the Southern California Bight by developing a regional, unified management strategy. Outputs from our model provide local and regional managers with valuable insights into the impacts of SLR at a variety of geographic scales. Partnership and collaboration in this effort includes 18 agencies across the state, which include academic and governmental research institutions, regulatory agencies, and non-profit organizations. Continued involvement in this effort required periodical presentations of the updates and findings of the work presented here, including:

- Monthly meetings with technical project team
- Quarterly update with scientific advisory team
- Quarterly updates with managers group consisting of all WRP partner agencies
- Quarterly discussions with WRP WAG, consisting of local wetland managers (i.e. end users)
- Collaboration with ESRI on development of outreach and information dissemination products
- Collaboration with TNC on their statewide vulnerability and conservation planning analyses
- Collaboration with Moss Landing Marine Labs on analogue efforts along the Central Coast

In addition, this analysis will comprise a chapter in the forthcoming RSU report. Our work will be presented at the Restore America's Estuaries (RAE) conference taking place in New Orleans, LA in December 2016. We also plan to publish our findings in a yet to be determined peer-reviewed journal.

Future Directions

More work is needed to improve the SLR response assessment presented here. In general, we hope that future iterations of our assessment will decrease regional data gaps. In addition, we plan to include a more robust uncertainty analysis which incorporates both qualitative confidence levels of data availability, as well as quantitative confidence intervals. To do so, we will conduct a sensitivity analysis on the inputs for each model component using either a book-end or Monte Carlo approach.

Our model can also be used to inform ongoing SCWRP RSU efforts in setting regional goals and planning management strategies. Potential applications include running the model for a variety of future management options for a given site and determining how each action will change the vulnerability. There a number of management strategies that can be incorporated into our model including the acquisition of upland transition zones, sediment augmentation and estuary mouth management.

Habitat change index (HCI) scores can also be adjusted based on how constrained a system is by human infrastructure. This is currently not incorporated into our model, but we plan to produce a constraint metric which reflects topography, impervious surfaces, road density, land use and population in the areas surrounding each site. This would improve our model for a region where human influence on coastal systems is paramount, and SLR vulnerability is equally determined by anthropogenic factors as well as climate change.

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