Assessment of Sea Level Rise Vulnerability for Southern California Coastal Estuaries

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September 2016
University of Southern California Sea Grant Traineeship Program Report

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) vulnerability are needed at scales that are meaningful to both site managers and regional regulators.

This project aims to improve our understanding of regional SLR vulnerability by assessing site-specific SLR response. We quantify SLR response as the change in wetland habitat composition arising from changes to elevation and inundation. For the purpose of this effort, "vulnerability" is used to describe the amount of divergence a system is expected to undergo from its current state. 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, vertical land motion, 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 vulnerability.

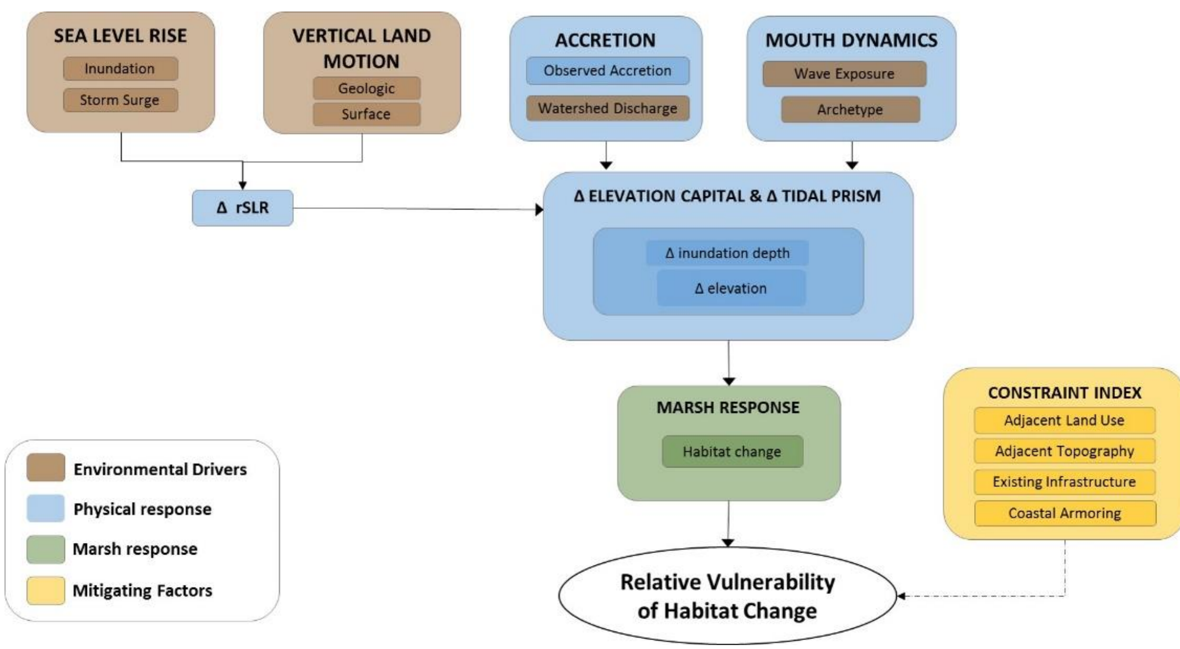

Figure ES1. The conceptual model for the vulnerability assessment showing the drivers and responses that are important in determining SLR response in the Southern California Region.

Our assessment was conducted for 104 systems throughout the Southern California Bight which range in size, structure and initial habitat composition. Overall, we found that for the 2050 SLR scenario, 35% of the coastal systems are vulnerable (scores > 3). For the 2100 SLR scenario, the number of vulnerable systems increased to 68% (Figure ES2). There was an increase of 29 systems that had high vulnerability scores from 2050 to 2100. We found that the most vulnerable systems for 2050 are the large perennially open lagoons, indicating that this archetype is at risk for drastic conversion of marsh habitats to subtidal and mudflat habitats. By 2100, the small creeks and small lagoons will be among the most vulnerable systems. The highest concentration of vulnerable sites is in Santa Barbara county where small lagoons are common.

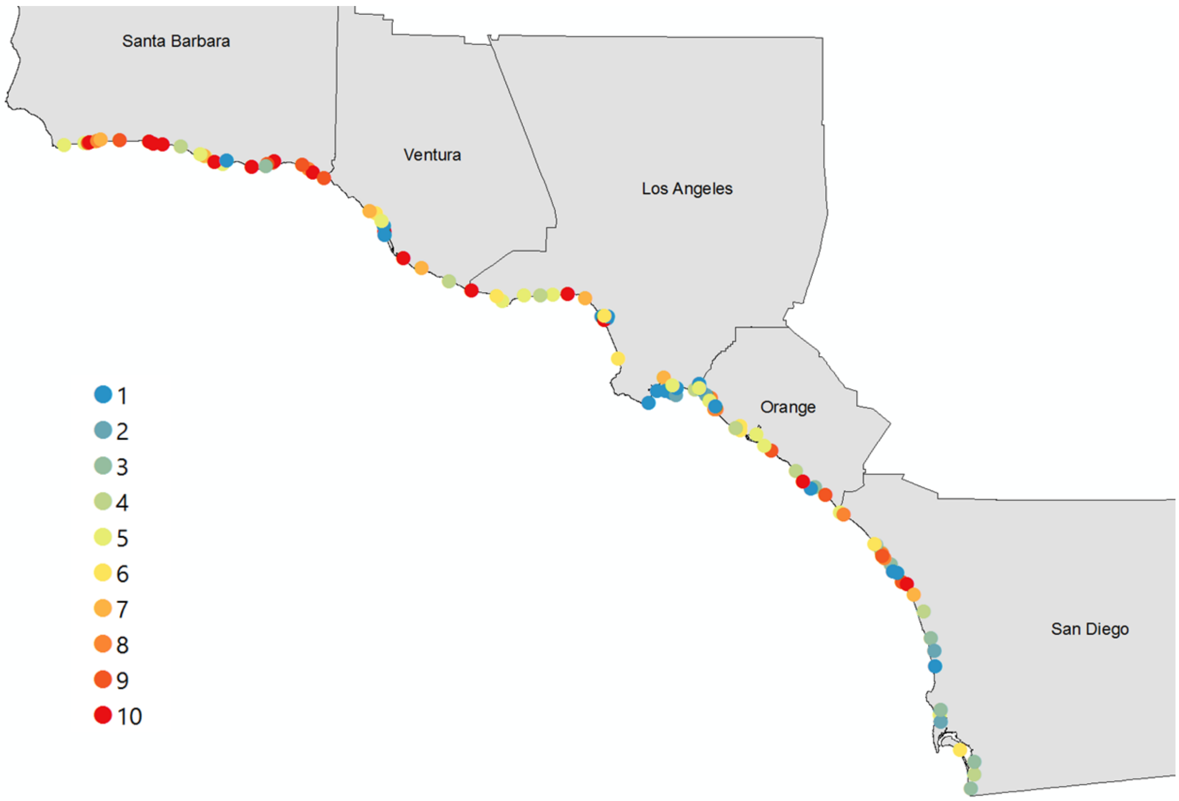

Figure ES2. 2100 SLR vulnerability for the 104 study sites throughout the Southern California region.

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 vulnerability. Our regional vulnerability 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 vulnerability 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 vulnerability 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 vulnerability 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. These results can be used to support regional planning efforts by providing a regional screening level assessment of SLR effects and vulnerability. Ultimately, this vulnerability 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 developed a conceptual model which reflects the environmental drivers, geophysical processes and biological feedbacks that are important in determining SLR vulnerability in this region (Figure 1). Here, we simplify this complex interaction into several factors which affect a system's ability to respond to SLR. Components that were ultimately incorporated in the model were determined by data availability, as well as expert knowledge of factors controlling SLR response in the region. Our basic approach uses the summation of these components to estimate potential changes in marsh elevation and water levels, which impact marsh hypsometry. This allows us to assess vulnerability as a relative change in habitat composition for a given system.

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

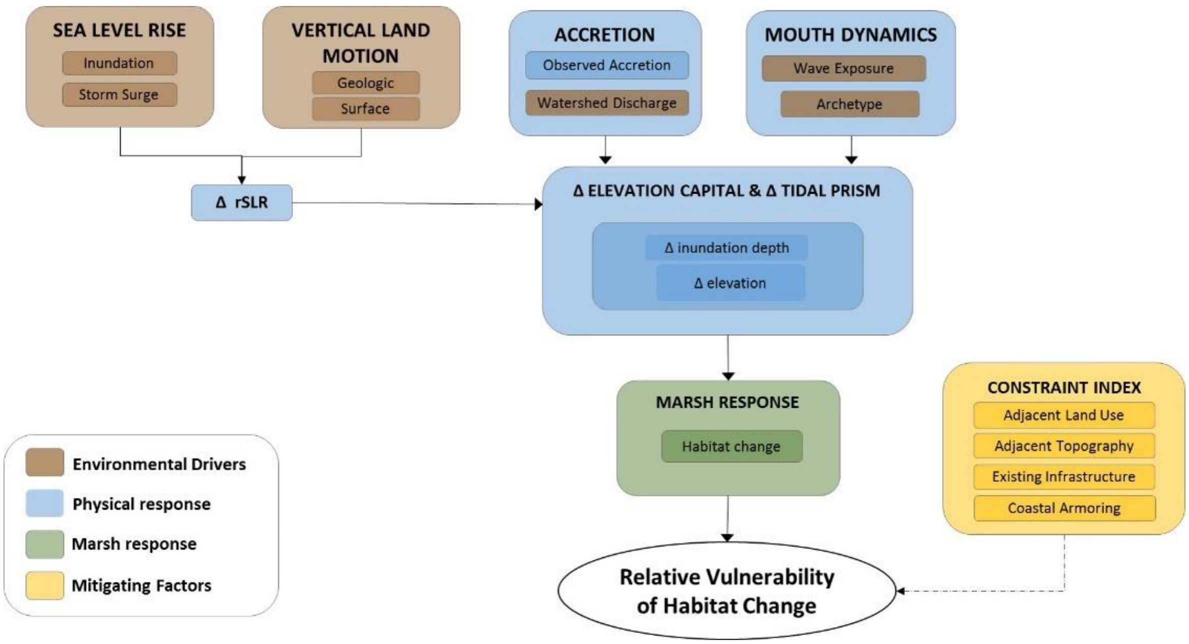

Figure 1. The conceptual model for the vulnerability assessment showing the drivers and responses that are important in determining SLR response in the Southern California Region.

Table 1. Data inputs for the vulnerability 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 vulnerability 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

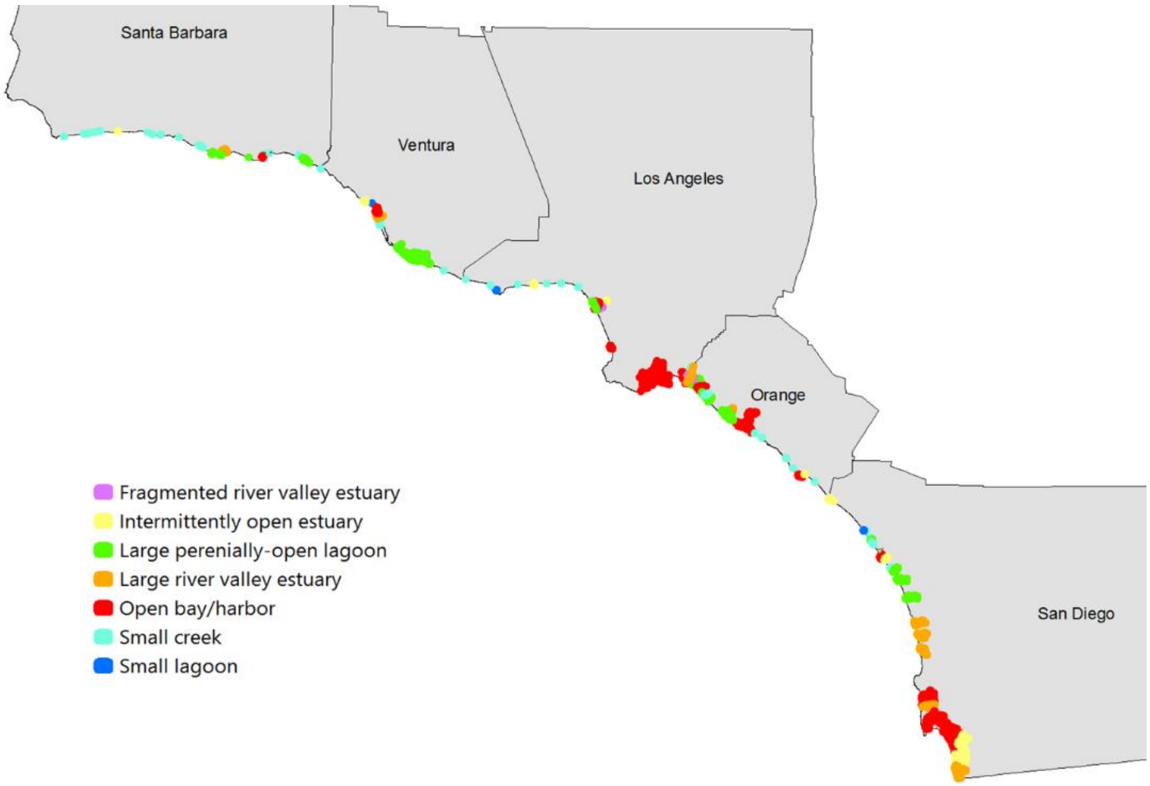

Figure 2. Focal estuaries in the Southern California region. Symbology indicates archetypes or groupings of estuaries based on similar characteristics. Estuaries sizes are expanded for visualization and are not to scale.

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 vulnerability 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.

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 vulnerability 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 vulnerability 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 (Ω= 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).

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.

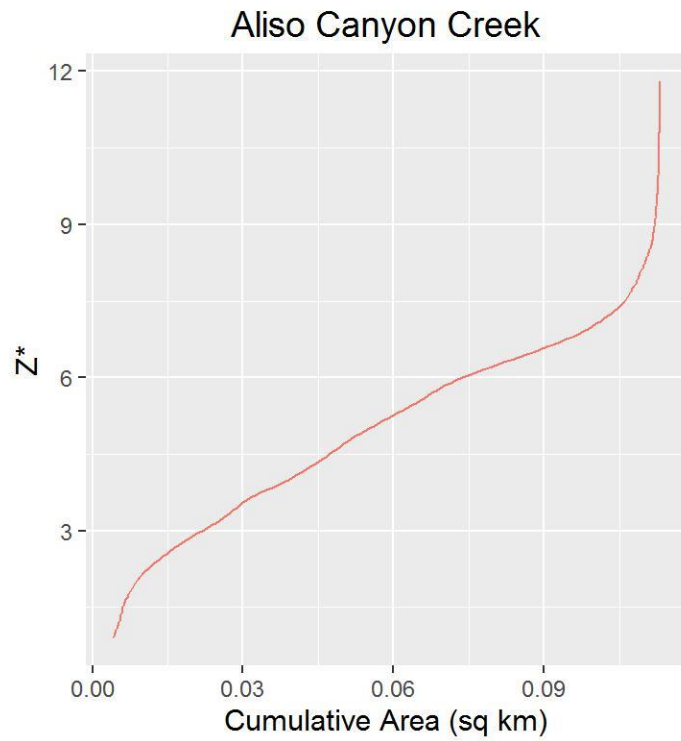

Figure 3. Hypsometric curve developed for Aliso Canyon Creek using the NOAA-CC LiDAR DEM.

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 (Δη) 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).

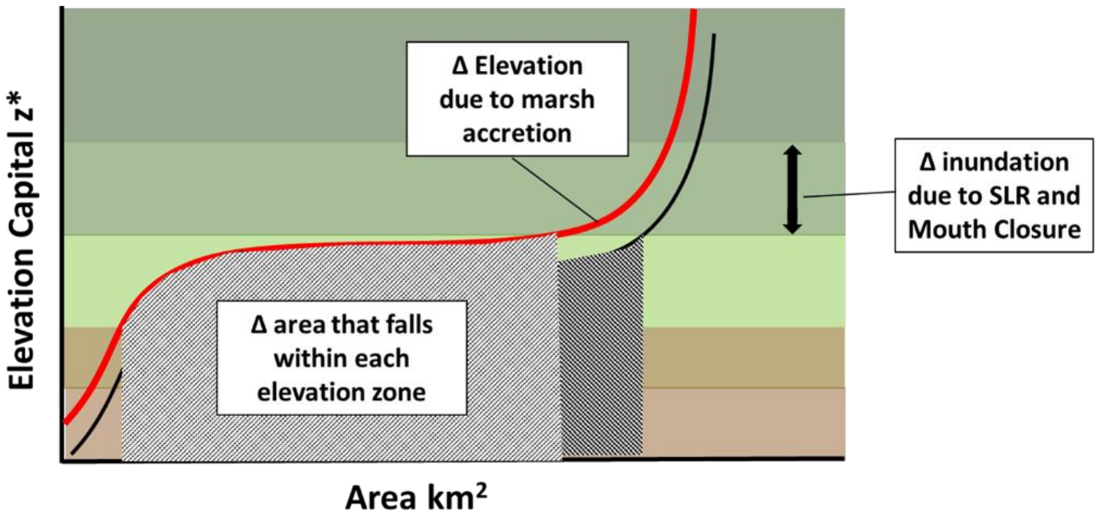

Figure 4. Conceptual diagram for integrated changes in elevation and water level with marsh hypsometry. Curved lines represent current (black) and future (red) marsh hypsometry which have been altered by changes in elevation. Background colors (brown – green) represent different marsh zones. Double-ended 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.

Vulnerability Scoring

We developed a method for scoring relative habitat change for each estuary. First, we summed the areas within each system that fall into several marsh zones (subtidal, intertidal mudflats, low-, mid-, and high-marsh). These calculations were performed for current, 2050 and 2100 scenarios and converted to percentage of the total system area. Absolute change in % area was then calculated for each marsh zone in each system for 2050 and 2100 compared to the current baseline. Change in percent area was summed across all marsh zones for a given system. Total percent change was normalized by the maximum total percent change for 2050 and 2100. Raw scores were then converted to a 1 – 10 scale by multiplying normalized percent change by 10. Resultant vulnerability scores reflect the change in habitat composition driven by SLR for 2050 and 2100 compared to the 2016 baseline. Vulnerability score were grouped into the following qualitative bins: low (<3.33), medium (>3.33,<6.66) and high (>6.66) vulnerability.

Uncertainty Analysis

Sources of uncertainty in our vulnerability analysis include SLR and VLM projections, accretion data availability, mouth dynamics assumptions, vertical datum accuracy, and the archetype-based extrapolation of data throughout the region. To address the uncertainty associated with data availability in this region, we assigned uncertainty scores for each of the key model inputs (Table 6). These scores reflect the origin of input data and were assigned for rSLR, accretion and mouth dynamics model components.

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.

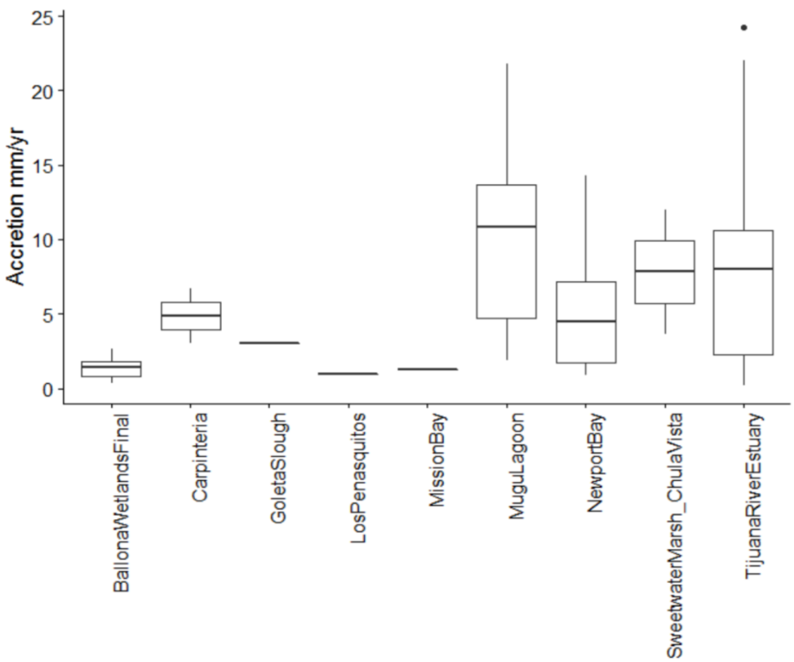

Figure 5. Boxplot of accretion estimates for a subset of estuaries in Southern California.

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 associated 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

Our model produced estimates of habitat change with SLR for 2050 and 2100 for each of the 104 systems. Here we show the potential changes habitat composition aggregated by archetypes.

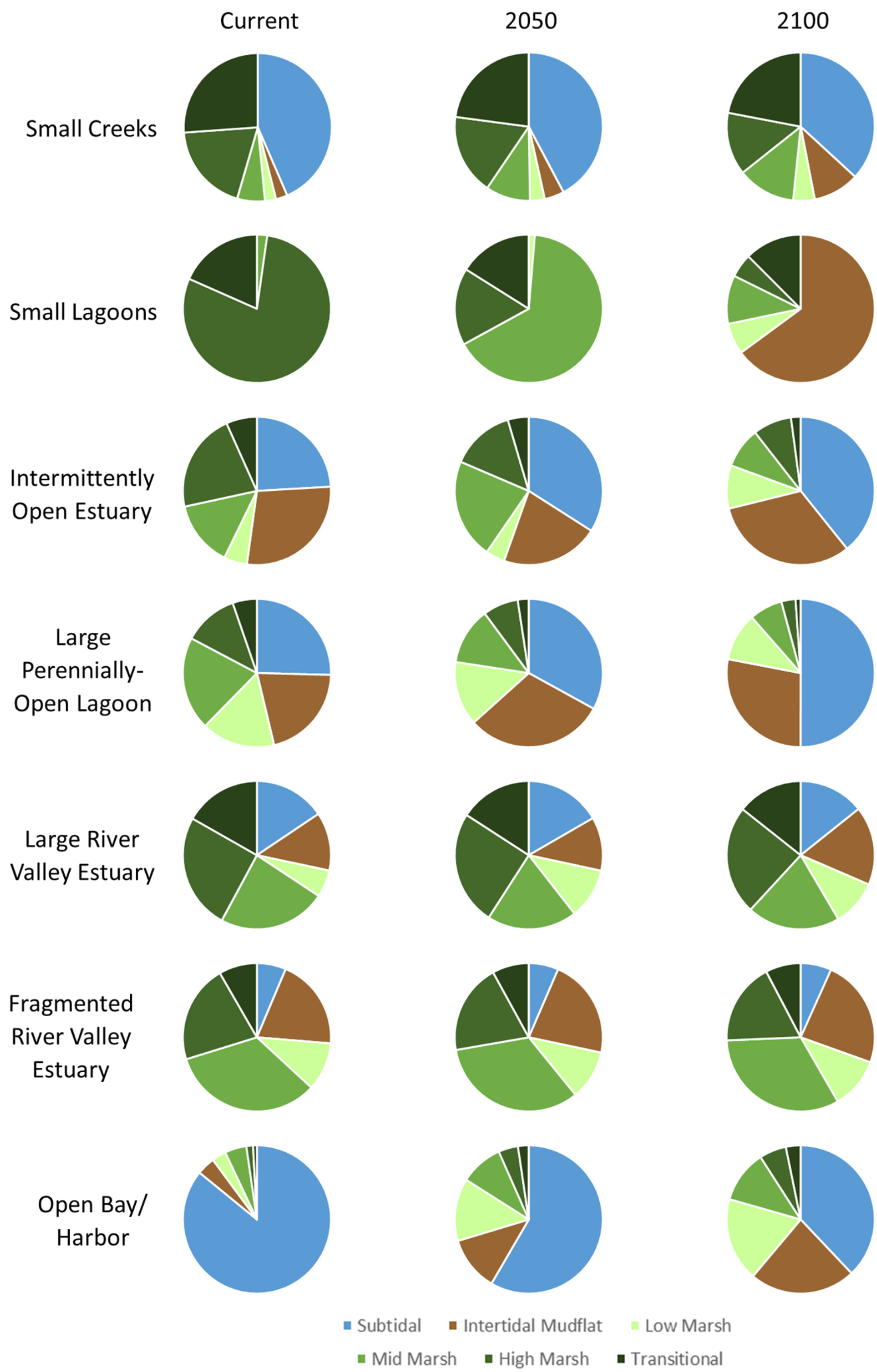

Figure 6. Predicted changes to habitat composition from current to 2050 and 2100 SLR scenarios.

Table 10. Predicted changes in percent area for each marsh zone. Baseline for comparison is 2016.

	% Change in Area	
Marsh Zone	**2050**	**2100**
Subtidal	- 10.9%	- 9.2%
Intertidal Mudflat	+ 6.2%	+ 10.2%
Low	+ 3.4%	+ 3.0%
Mid	+ 0.9%	-2.5%
High	+ 0.3%	-1.6%

Vulnerability Scores

Overall, we found that for the 2050 SLR scenario, 68 systems had low vulnerability (<3.33), 28 systems had medium vulnerability (>3.33, <6.66), and 8 systems had high vulnerability (>6.66) (Figure 6). For the 2100 SLR scenario, 33 systems had low vulnerability, 34 systems had medium vulnerability, and 37 systems had high vulnerability (Figure 7). There was an increase of 29 systems that had high vulnerability scores from 2050 to 2100. We found that the most vulnerable systems for 2050 are the large perennially open lagoons. By 2100, the small creek and small lagoon archetypes will be among the most vulnerable systems. In addition, we found that the Santa Barbara subregion has the highest concentration of vulnerable sites for both 2050 and 2100.

Figure 7. SLR vulnerability for 2050

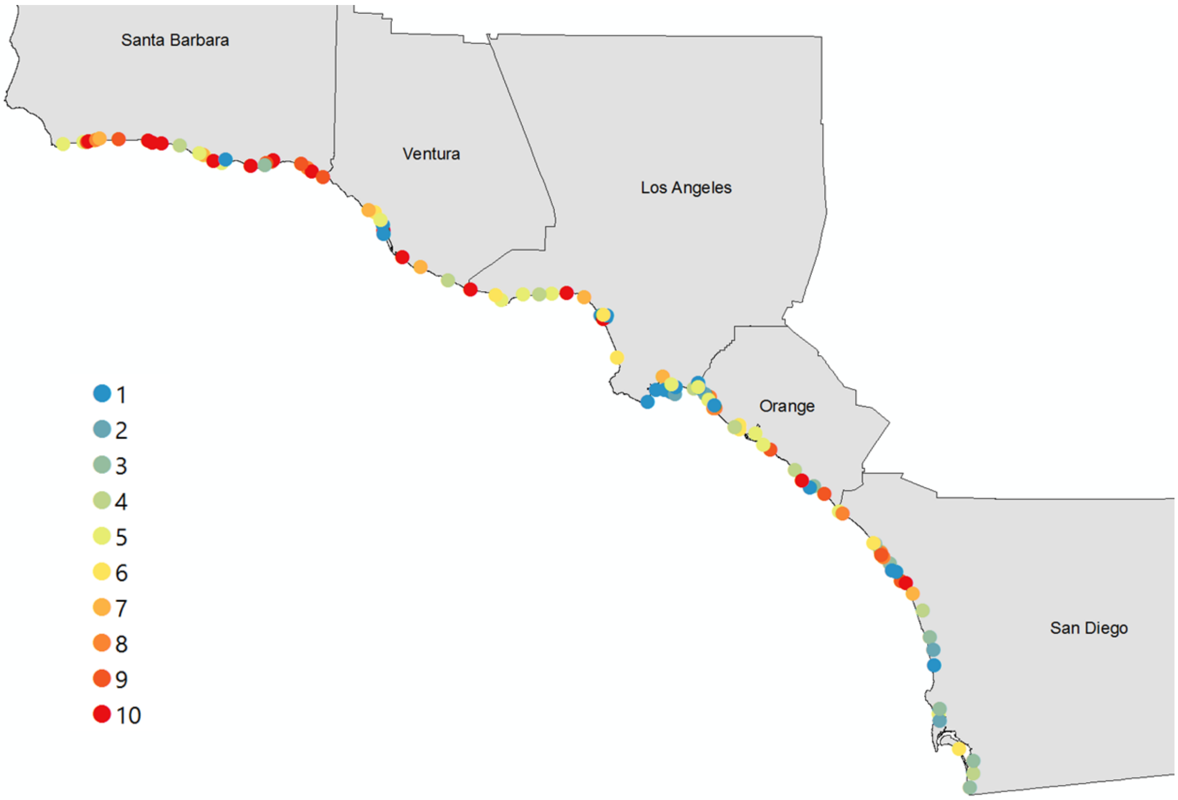

Figure 8. SLR vulnerability for 2100.

Uncertainty

Our uncertainty analysis reflects the availability of data throughout the region. Sites with low uncertainty were data-rich, while sites with high uncertainty were lacking data. We found that 42 sites have high uncertainty, 52 sites have medium uncertainty, and 10 have low uncertainty.

Figure 9. Uncertainty scores reflecting data availability in the region.

Significance

Through this process, we produced a quantitative estimate of relative vulnerability 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 vulnerability to 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 vulnerability at a given site. Because this effort uses site estimates of habitat change to determine future vulnerability, it strikes the balance between large-scale vulnerability assessments and site-specific marsh response models. Regional vulnerability 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 vulnerability 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 vulnerability 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.

Vulnerability 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.

Acknowledgements

Funding for this work was provided by grants from the USC Sea Grant Trainee Program and the U.S. Fish and Wildlife Service Landscape Conservation Cooperative (LCC) program. This work was conducted as part of the Southern California Wetland Recovery Project's (WRP) Regional Strategy Update and benefited from input and guidance from the WRP's Science Advisory Panel. Cheryl Doughty was also supported by a part-time graduate researcher position at the Southern California Coastal Water Research Project (SCCWRP). Special thanks to Jeremy Lowe, Carolyn Doehring and Heather Dennis at the San Francisco Estuarine Institute (SFEI), Megan Cooper and Evyan Borgnis at the CA Coastal Conservancy, and John Largier at UC Davis Bodega Bay Marine Lab for their technical input and guidance. Finally, we would also like to thank Phyllis Grifman and Ruth Dudas at the USC Sea Grant Program for their continued support.

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