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Evaluating regional resiliency of coastal wetlands to sea level rise through hypsometry-based modeling

Cheryl L. Doughty[1] | Kyle C. Cavanaugh[1] | Richard F. Ambrose[2] | Eric D. Stein[3]

[1]Department of Geography, University of California, Los Angeles, California

[2]Department of Environmental Health Sciences, Institute of the Environment and Sustainability, University of California, Los Angeles, California

[3]Biology Department, Southern California Coastal Water Research Project, Costa Mesa, California

Correspondence
Cheryl L. Doughty, Department of Geography, University of California, Los Angeles, Los Angeles, CA.
Email: cdoughty@ucla.edu

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Abstract

Sea level rise (SLR) threatens coastal wetlands worldwide, yet the fate of individual wetlands will vary based on local topography, wetland morphology, sediment dynamics, hydrologic processes, and plant-mediated feedbacks. Local variability in these factors makes it difficult to predict SLR effects across wetlands or to develop a holistic regional perspective on SLR response for a diversity of wetland types. To improve regional predictions of SLR impacts to coastal wetlands, we developed a model that addresses the scale-dependent factors controlling SLR response and accommodates different levels of data availability. The model quantifies SLR-driven habitat conversion within wetlands across a region by predicting changes in individual wetland hypsometry. This standardized approach can be applied to all wetlands in a region regardless of data availability, making it ideal for modeling SLR response across a range of scales. Our model was applied to 105 wetlands in southern California that spanned a broad range of typology and data availability. Our findings suggest that if wetlands are confined to their current extents, the region will lose 12% of marsh habitats (vegetated marsh and unvegetated flats) with 0.6 m of SLR (projected for 2050) and 48% with 1.7 m of SLR (projected for 2100). Habitat conversion was more drastic in wetlands with larger proportions of marsh habitats relative to subtidal habitats and occurred more rapidly in small lagoons relative to larger sites. Our assessment can inform management of coastal wetland vulnerability, improve understanding of the SLR drivers relevant to individual wetlands, and highlight significant data gaps that impede SLR response modeling across spatial scales. This approach augments regional SLR assessments by considering spatial variability in SLR response drivers, addressing data gaps, and accommodating wetland diversity, which will provide greater insights into regional SLR response that are relevant to coastal management and restoration efforts.

KEYWORDS
archetypes, climate change, coastal management, coastal wetlands, habitat change, hypsometry, vulnerability

1 | INTRODUCTION

Sea level rise (SLR) and its impacts to coastal areas have been documented throughout the world (Wong et al., 2014) and are predicted to worsen in the coming decades (Church et al., 2013). Global sea levels are increasing, yet our ability to predict SLR impacts to coastal systems is complicated by variability in the factors driving SLR and system response (Stammer, Cazenave, Ponte, & Tamisiea, 2013). The climatic, geologic, and hydrologic processes that contribute to the relative SLR experienced along a coast are highly variable in time and space (Cazenave & Le Cozannet, 2013). At continental or regional scales, these factors include the gravitational properties and movement of the earth's surface (Bamber & Riva, 2010; King et al., 2012; Riva, Bamber, Lavallée, & Wouters, 2010), and the circulation and volume of the ocean due to shifting surface winds, melting land ice contributions, and the thermal expansion of seawater (Church et al., 2013; Rhein et al., 2013). Subregional and local processes include tectonics, coastal geomorphology, and hydrology (Behrens, Brennan, & Battalio, 2015; Cahoon et al., 2006; Rich & Keller, 2013).

The variability associated with these SLR drivers makes it difficult to predict how SLR will affect coastal wetlands across broad spatial scales. The fate of individual wetlands will be determined by local rates of SLR, local topography, and wetland morphology, but SLR response will be further mediated by biogeomorphic feedbacks among inundation, plant growth, organic matter accretion, and sediment deposition (Kirwan et al., 2010; Mueller, Jensen, & Megonigal, 2016). Biogeomorphic feedbacks control vertical accretion and wetland elevation and so are important in determining response to SLR (Morris, Sundareshwar, Nietch, Kjerfve, & Cahoon, 2002). These complex interactions determine which wetland sites may be able to keep pace with rising sea levels and which will not (Kirwan, Temmerman, Skeehan, Guntenspergen, & Faghe, 2016). Consideration of the spatial variability in both SLR drivers and SLR response mechanisms is necessary to compare SLR effects across individual wetlands.

The ability of SLR assessments to accommodate spatial variability in driver and response mechanisms is determined by data availability and computational modeling capability. Large-scale assessments, that is, regional, national, continental, or global (Gornitz, 1991; Klein & Nicholls, 1999; Spencer et al., 2016), are purposefully designed to provide synoptic insights into SLR impacts by lowering computational expense and simplifying representation of the processes driving SLR (Fagherazzi et al., 2012). Large-scale assessments are often conducted at spatial resolutions of 1 km^2 or greater (Passeri et al., 2015) or characterize sites or segments of coastline under a broad classification scheme (Lentz et al., 2016; Nicholls, 2004; Spencer et al., 2016). Using broad classes and coarse spatial scales can obscure entire systems and some wetland types within a region, potentially biasing overall conclusions of wetland change.

Conversely, fine-scale assessments are better equipped to capture spatial variability at local scales (meters to 10 s of meters) within individual wetlands. For example, process-based models such as the Wetland Accretion Rate Model for Ecosystem Resilience (WARMER) (Swanson et al., 2014), the Sea Level Affecting Marshes Model (SLAMM) (Clough, Park, Propato, Polaczyk, & Fuller, 2012), the Marsh Equilibrium Model (MEM; Morris et al., 2002), and the integrated Hydro-MEM (Alizad, Hagen, Morris, Bacopoulous et al., 2016) incorporate biogeomorphic feedbacks associated with inundation, plant growth, organic matter accretion, and sediment deposition. When used in conjunction with site-specific data on vegetation and/or elevation, the results can provide insights into SLR response at high spatial resolutions (Alizad, Hagen, Morris, Medeiros et al., 2016; Thorne et al., 2018). However, such comprehensive modeling efforts are often specific to a single wetland site and are difficult to replicate over broad geographic ranges for a variety of wetlands due to time and resource constraints.

The limitations inherent to large-scale and fine-scale assessments have led to geographical gaps in our understanding of SLR response. The availability and accessibility of relevant data, as well as bias in assessment design caused by stakeholder incentives and resource availability, also contribute to these gaps (Preston, Yuen, & Westaway, 2011). As a result, modeling efforts are normally conducted in well-studied and data-rich regions, wetland types, or specific sites. The remaining, less-studied areas warrant attention, given that SLR response is highly context-dependent. However, coastal wetland types with similar origin, geomorphology, dynamics, sediment balance, biogeochemistry, and ecology will respond similarly to SLR and can therefore be grouped into a typology to provide a framework for the transfer of knowledge from data-rich to data-poor wetlands (Vafeidis et al., 2008). Here, coastal wetland typology reflects general classes such as small deltas, tidal systems, lagoons, large rivers, estuaries, and bays (Dürr et al., 2011). The inclusion of all wetlands and wetlands types would provide a more holistic regional perspective on SLR response that is required by both local and regional management efforts to ensure the resiliency of coastal ecosystem networks in the future (Gilmer, Brenner, & Sheets, 2012; Stralberg et al., 2011).

This study aims to develop a nested SLR response assessment model that can be used to estimate SLR-induced habitat change across large geographic regions containing diverse wetland types. Our work augments current regional SLR assessments, bridging the gap between coarse regional and detailed site-specific models by addressing (a) the spatial variability of SLR drivers and response, (b) the ability to assess wetlands of different sizes and typologies using a common approach, and (c) the need to accommodate differences in data availability across sites. We conducted our assessment in the southern California region, which has a diverse range of wetland typologies representing many of the wetland types found globally, including Mediterranean areas, which have often been overlooked by previous SLR assessments. In addition, data availability varies a great deal for wetlands across southern California. Model outputs provide relative estimates of change in habitat composition for local (site specific) and regional scales, which are necessary to inform regional SLR adaptation efforts in southern California. Our goal is to estimate overall wetland losses for the region due to future SLR and explore how relative losses would vary among wetland types. The modeling approach is also highly relevant to other coastal regions throughout the world where predicting future SLR response is complicated by habitat heterogeneity and limited data availability.

2 | MATERIALS AND METHODS

2.1 | Model overview

We developed a rule-based model that quantifies SLR-driven habitat conversion caused by the combined effects of SLR, accretion, and changes in water levels due to estuary mouth dynamics (Figure 1). The model uses hypsometric data, the measure of land elevation in relation to sea level, as a standardized basis for quantifying habitat change within individual wetlands across a region. Data gaps are addressed by incorporating the best available data at local to regional scales and by developing a wetland archetype framework to extrapolate data across similar wetland types. We conducted our assessment in southern California and parameterized our model using data collected for the 105 coastal wetlands found in the region (Table 1). The model was executed for two SLR projections, 0.6 m by 2050 and 1.7 m by 2100, based on regional guidelines (California Coastal Commission, 2015; Griggs et al., 2017; National Research Council, 2012). Wetlands were constrained to their current boundaries, that is, no upland migration was allowed, due to the uncertainties associated with predicting wetland migration (Anisfeld, Cooper, & Kemp, 2017). Modeling was conducted in R (R Foundation for Statistical Computing, Vienna, Austria).

2.2 | Regional background

Southern California is emblematic of many coastal regions worldwide in that it contains a large number of wetlands of various sizes and types. Historically, this region contained over 300 coastal wetlands occupying approximately 19,591 ha of wetland habitat (Stein et al., 2014). Since ca. 1800, close to 50% of wetland areas in this region have been converted to open water or non-estuarine habitat, that is, developed, agricultural, or open space land uses (Stein et al., 2014). The existing wetland remnants are geographically isolated and constrained due to human modification of the landscape to serve urban development, military uses, industrial and agricultural expansion, recreation, and tourism (Zedler, 1982).

2.3 | Wetland archetype classification

We have grouped wetlands in this region into archetypes to facilitate extrapolation of data between wetlands to fill the gaps in our knowledge of the region. Archetypes represent wetlands with similar physical structure and ecosystem drivers that are expected to react in similar ways to SLR. To develop the archetypes, we compiled data on the physical structure and processes for each wetland in the region; these variables fell into five general categories including catchment properties, wetland dimensions (e.g., size and slope), proportion of subtidal to intertidal areas, inlet dimensions and area, and wetland volume and capacity. Of the 105 wetlands, 46 had sufficient data to be included in a K-Means cluster analysis where cluster numbers ranging from 4 to 8 were tested to maximize separation and minimize misclassification rates.

The resulting archetypes were further refined using a discriminant function analysis to identify the key predictor variables that generated the greatest accuracy of classification. Key predictors included wetland area, erosion area (area/depth), slope from mouth to head, integrated slope, mouth elevation, mean mouth width, total

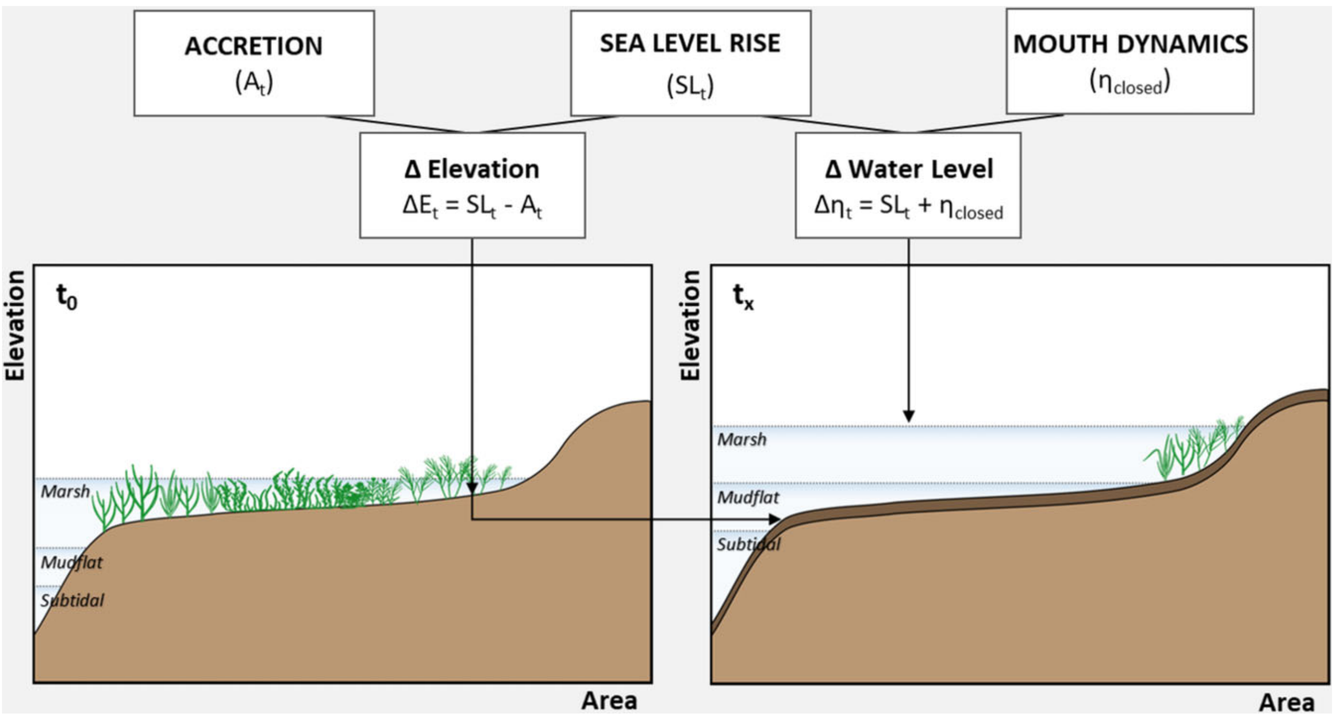

FIGURE 1 Conceptual diagram of the SLR response habitat change model. Model drivers (sea level rise, accretion, and mouth dynamics) represent the processes inducing change in water levels and elevation. Elevation change (ΔE_t) raises the current marsh hypsometry (t_0, light brown) to future hypsometry ((t_x), dark brown). Water level changes ($\Delta\eta_t$) alter the marsh zones (subtidal, mudflat, and vegetated marsh) delineated by elevation (horizontal lines). Habitat change is calculated as the difference in area under the curve for each marsh zone under current and future conditions

TABLE 1 Data inputs required to inform and parameterize the SLR habitat change model

Input	Scale	Value (units)	Source
Relative sea level rise			
SLR 2050 Projection	Regional	12.2 (mm/year)	National Research Council (2012) and Griggs et al. (2017)
SLR 2100 Projection	Regional	16.6 (mm/year)	National Research Council, (2012) and Griggs et al. (2017)
t_0, t_1, t_2		2016, 2050, 2100 (year)	
Accretion			
Measured accretion ± Error	Site, Archetype	mm/year	SCCWRP Literature Review
Mouth dynamics			
Daily water levels	Subregional	m	NOAA
Daily wave height, period, direction	Subregional	m, s, degrees	CDIP
Watershed Runoff estimates	Subregional	m^3/s	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	Subregional		SCCWRP/SFEI Literature Review
LiDAR-derived DEM	Regional	m^2	2009–2011 CA Coastal Conservancy Coastal Topobathy Project
Estuary hypsometry	Site	km^2/z^*	This study

Note. NRC, National Resource Council; SCCWRP, Southern California Coastal Water Research Project; NOAA, National Oceanic and Atmospheric Association; CDIP, Coastal Data Information Program; SFEI, San Francisco Estuary Institute.

area inundated at spill height, percent wetland at low tide, and total percent subtidal. Finally, we mapped habitat data from the National Wetlands Inventory (NWI) and the Classification and Assessment with Landsat of Visible Ecological Groupings (CalVeg) system onto the clusters to produce habitat associations for each archetype. The validity of the resulting archetypes was tested by performing a bias analysis on the key predictor variables and the remaining 59 systems not included in the cluster analysis. Archetypes were also qualitatively assessed by regional experts who were allowed to add modifiers to each wetland based on mouth armoring, mouth migration potential, and/or the presence of engineered channels.

The resulting archetypes include small creeks, small lagoons, intermediate estuaries, large lagoons, large river valley estuaries, fragmented river valley estuaries, and open bays and harbors (SCWRP, 2018). Intermediate estuaries, commonly referred to as intermittently opening and closing estuaries or bar-built estuaries, are noteworthy because they represent a significant component of coastal wetlands in southern California (Zedler, 1996). Each of the 105 wetlands in the analysis was classified as one of the seven archetypes (Figure 2).

2.4 | Model components

2.4.1 | Sea level rise

Sea level projections for the Los Angeles area range from 12.7 to 60.8 cm for the year 2050 and from 44.2 to 166.5 cm for 2100 (National Research Council, 2012). We selected the maximum of the projected SLR ranges for 2050 and 2100 (0.6 m and 1.7 m, respectively). We converted these projected levels of inundation to SLR rates (mm/year) by dividing by the difference in time period between our modeling time points (2050, 2100) and the reports' baseline (2000), resulting in SLR rates of 12.2 mm/year for 2050 and 16.6 mm/year for 2100. A regional default value for SLR was selected to provide a standardized baseline for comparison across wetlands and also to align with ongoing regional efforts (SCWRP, 2018).

2.4.2 | Accretion

Empirical estimates of accretion in coastal wetlands in southern California were obtained through a review of published literature (Supporting information Table S1). The records included in the analysis were derived from radiocarbon dating, radiocesium dating, and paleoenvironmental soil core analysis of pollen and cover periods ranging from ~30 to 10,000 years. Accretion estimates were standardized to millimeters per year (mm/year). Records reflecting short-term (<10 year) accretion and episodic sediment deposition from storms were excluded from the analysis to ensure that model inputs reflect long-term accretion rates on temporal scales comparable to SLR projections.

Because of the limited availability of published accretion records specific to this region (see *Results*), we used the archetype framework to extrapolate data from well-studied sites to data-poor sites. When empirical data were available for a given wetland, we used site-wide averages across all intertidal marsh zones as accretion

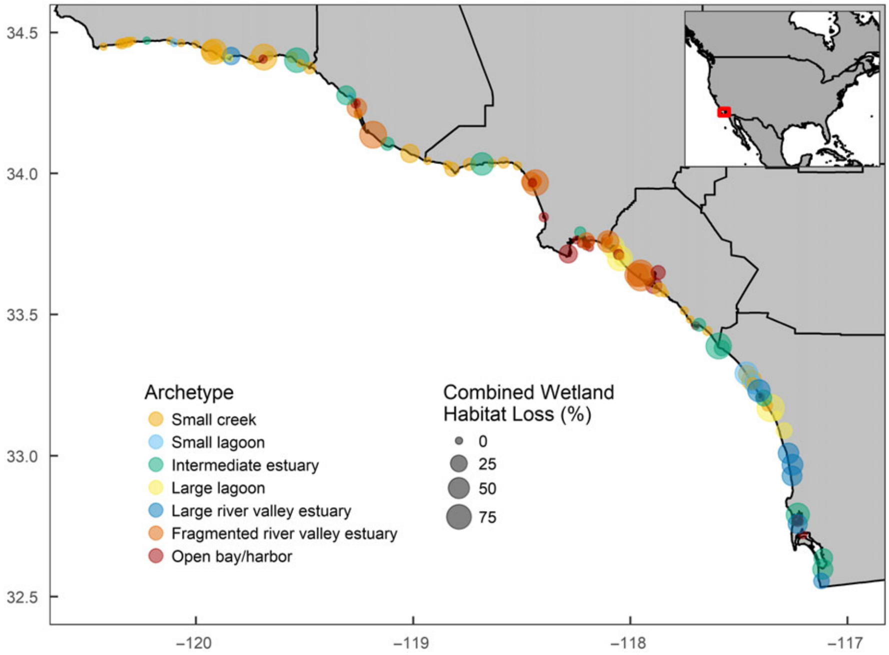

FIGURE 2 Archetype classification and projected percent difference in combined wetland habitat (vegetated marsh and unvegetated mudflat) for individual wetland sites for 1.7 m SLR (2100) in the southern California region

input. For wetlands lacking empirical data, we calculated archetype accretion estimates from our literature review by aggregating records first by marsh zone within a site, then by site, and finally by archetype to provide inputs for similar sites specific to this region (Table 2). Accretion rates for small creeks and lagoons were set to 0 mm/year because of a lack of empirical data and to avoid additional uncertainty associated with modeling fluvial sediment discharge. Contributions to accretion from both allochthonous and autochthonous inputs are relatively low for these systems given the variable precipitation patterns in southern California, small catchment sizes and the small vegetated areas associated with these systems.

TABLE 2 Archetype accretion rates estimated from the literature review

		Accretion ± *SD* (mm/year)		
Archetype	**Zone**	**By Zone**	**Total**	***n***
Small creek		No Data		0
Small lagoon		No Data		0
Intermediate estuary	Low	3.63	3.63	1
Large lagoon	Mid	12	9.55 ± 9.40	4
	High	21.8		
Large river valley estuary	Low	4.8	4.68 ± 2.94	5
	Mid	4.83 ± 5.49		
Fragmented river valley estuary	Low	1.38	1.2 ± 0.65	3
	Mid	0.49		
Open bay/harbor	Low	2.83	3.84 ± 2.32	5
	Mid	3.24 ± 2.82		
	High	7.13		

2.4.3 | Mouth dynamics

We included an estuary mouth dynamics component in our assessment to address the intermittently opening and closing state of estuary mouths common to southern California and throughout the world (Rich & Keller, 2013). 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 the height and position of terminal estuary bars. As estuary bars rise with sea level, the flood risk associated with estuary closure is expected to worsen (Behrens et al., 2015). These changes can in turn affect extent and duration of inundation and therefore habitat distributions. To inform our model, we conducted a separate modeling analysis which includes a closure index (PWA, 2003; Williams & Cuffe, 1995) and a water balance model (Behrens, Bombardelli, Largier, & Twohy, 2013; Behrens et al., 2015).

These analyses were conducted to estimate the probability of changing mouth state and the subsequent changes to estuary water levels for individual sites. We hypothesize that increased sea levels will increase the frequency of estuary closure, inducing a shift in dominant mouth state, and will ultimately increase water levels within a site. To test our hypothesis, we created a synthetic daily time series for current, 2050, and 2100 sea levels using local NOAA tide level data, Coastal Data Information Program (CDIP) water level and wave data, and Southern California Coastal Water Research Project (SCCWRP) coastal watershed runoff data for the last 20 years (Table 1). To predict future closure indices, we manipulated wave and tidal inputs to reflect SLR projections of 0.6 m for 2050 and 1.7 m for 2100.

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):

$$S = P_w / P_t \tag{1}$$

$$P_w = 0.5 \rho g H_s C \tag{2}$$

$$P_t = (\gamma h_T)/b \times (\Omega/T + Q) \tag{3}$$

where P_w is wave power, ρ is the constant 1 kg/L 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 1,000 kg unit weight of water, h_T is the tidal range, b is the estuary 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 likely overpredicts mouth closure; however, we wanted to calculate the percent increase in predicted closure associated with SLR and the hypothetical changes to system 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 systems where data were available. We estimated daily system 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 system water level (η) would track mean sea level (MSL):

$$\eta = \text{MSL} \tag{4}$$

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

$$\eta_{t+1} = \eta_t + Q_{\text{net}}/\text{Area}_\eta \tag{5}$$

$$Q_{\text{net}} = Q_{\text{river}} - Q_{\text{evap}} \tag{6}$$

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

Seventy of the 105 sites in the region were considered in our mouth dynamics analysis because they did not have openly engineered estuary mouths. Of these, 36 sites had sufficient data and fall within several archetypes: small creeks, small lagoons, intermediate estuaries, large perennially open lagoons, and large river valley estuaries. For these sites, we calculated the percent of time that the site was expected to have high closure risk and the hypothetical changes to water levels when we assume the site is closed. Estimates of percent closure for 2050 and 2100 were added to estuary closure estimates for the present day and were then binned into the following categories: predominantly open (<40%), intermittently open/closed (>40%, <60%), and predominantly closed (>60%). Predominantly open sites received no additional changes in water level; intermittently open/closed sites received a dampened (0.5×) increase in water level; and predominantly closed sites received the full (1×) increase in water level associated with closed mouth state. The findings of our estuary mouth dynamics modeling analysis were aggregated by archetype to extrapolate to other systems with similar mouth dynamics characteristics but without sufficient data to be included in our mouth dynamics model.

2.5 | Wetland hypsometry

Hypsometric data were developed for the extent of each of the 105 wetlands using a digital elevation model (DEM) obtained from the 2009–2011 NOAA-CA Coastal Conservancy Coastal Lidar Project (Table 1). The DEM had a spatial resolution of 1 m^2 and horizontal and vertical error of 50 cm and 9.4 cm, respectively. DEM elevation relative to the North American Vertical Datum 1988 (z) was converted to z^*;

$$z* = \frac{z - \text{MSL}}{\text{MHHW} - \text{MSL}} \tag{7}$$

where z^* is the relative elevation within the tidal range calculated as a dimensionless ratio of elevation referenced to mean sea level (MSL) and mean high high water (MHHW) from the nearest NOAA tidal station (Swanson et al., 2014). z^* was used in order to standardize estimates of elevation changes across wetlands with varying tidal datums, elevations, and tidal ranges. Standard hypsometric curves providing information on wetland elevation relative to sea level were created by cumulatively summing the area that falls within z^* bins of 0.05.

2.6 | Changes in elevation and water level

Model components for SLR, accretion, and mouth dynamics were used to estimate changes in elevation and water level relative to our 2016 (t_0) baseline. Change in marsh elevation (ΔE) is determined by SLR and accretion:

$$\Delta E_t = SL_t - A_t \tag{8}$$

where SL_t is the change in sea level occurring by the year benchmark ($t_{1,2}$ = 2050, 2100) associated with the SLR projections 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 SLR and mouth dynamics:

$$\Delta\eta_t = SL_t + \eta_{\text{closed}} \tag{9}$$

where η_{closed} is the hypothetical change in water level when a site is assumed to be closed (see *Mouth dynamics*). ΔE_t and $\Delta\eta_t$ were estimated for all 105 wetlands for each SLR scenario.

2.7 | Habitat change

We used the estimated changes in elevation and water levels for 2050 and 2100, along with the hypsometric curves developed for each wetland to estimate areal changes in habitat occurring with SLR (Figure 1). Changes in elevation act upon the hypsometric curve itself, increasing the z^* values by ΔE (which has been converted from meters to z^* using the local tide datum), while keeping the total area of the marsh constant. Changes in water level are used to manipulate the z^* ranges that correspond to different wetlands habitats (Figure 1, supporting information Table S2). z^* ranges were informed by a synthesis of regional habitat-elevation data.

We calculated the area within three habitat classes (subtidal, unvegetated mudflat, and vegetated marsh) for each of the 105 wetlands under current sea levels and 0.6 m and 1.7 m SLR scenarios. We report the estimated changes to each habitat class and two metrics of wetland habitat loss. For the purpose of this study, wetland habitat loss reflects the combined loss of vegetated marsh and unvegetated mudflat areas, which assigns equal weight to these ecologically important classes and also aligns with ongoing regional efforts (SCWRP, 2018). First, we calculated percent change as the difference in wetland area between the existing area and the predicted area under the SLR scenario divided by the existing area (hereafter "percent change"). Second, we calculated percent difference as the decrease in the percent of wetland area for each site between the existing conditions and the predicted conditions under the SLR scenarios to produce an estimate of the relative decrease in the percent of wetland area (hereafter "percent difference"). We report percent loss using the "percent difference" approach in order to account for site sizes when comparing wetland loss among sites, archetypes, and the region.

2.8 | Uncertainty and sensitivity analyses

We conducted an uncertainty analysis to address errors of measurement and sensitivity to model parameters that may influence model outputs. We propagated errors (e.g., standard error, standard deviation, and 95% confidence intervals) associated with each of the model inputs through the model to determine the cumulative errors in the habitat change output. Model inputs considered include SLR rate, accretion rate, water level changes caused by mouth dynamics, and vertical error of the DEM. This error analysis provides us with a bookended range of potential habitat change for each site. We conducted a sensitivity analysis by modifying each input ±50% while leaving all other inputs unchanged. We modified DEM inputs by ±50 cm to test sensitivity to initial elevation for a broad, hypothetical range of vertical error feasible for digital elevation datasets. The sensitivity analysis provides an estimate of the percent change in habitat caused by modifying the inputs, which allows us to identify the importance of each input in predicting habitat change.

3 | RESULTS

3.1 | Regional accretion and estuary mouth dynamics

Investigation of regional accretion and mouth dynamics was conducted to inform the model and revealed inter-archetype differences that may cause differential response to SLR. Over 100 records of accretion rates were obtained for 10 wetland sites from 16 published sources, and of these, 58 records were suitable for analysis (Supporting information Table S1). Estimates of accretion per archetype based on the site-wide averages ranged from 1.2 ± 0.7 mm/year in fragmented river valleys to 9.5 ± 9.4 mm/year in large lagoons (Table 2). Differences in accretion between archetypes were useful for modeling but were not statistically significant given low sample sizes following data aggregation due to data limitations of this study (Table 2). Similarly, the mouth dynamics modeling analysis indicates that certain archetypes may experience increased likelihood of closure causing increased estuary water levels with future SLR (Table 3). For 2050, likelihood of closure was predicted to increase by 0%–13% with water levels increasing by an average of 34 cm and by 0%–48% with increases of 114 cm for 2100. Specifically, small creeks and lagoons are most at risk for increased closure, while larger sites with sufficient fluvial runoff are more likely to remain open. Although differences in increased likelihood of closure were detected, estimates of increased estuary water levels were similar

TABLE 3 Mouth dynamic analysis results

	0.6 m SLR		1.7 m SLR	
Archetype	Δ Likelihood of closure (%)	Δ Estuary water level (m)	Δ Likelihood of closure (%)	Δ Estuary water level (m)
Small creek	+13%	0.43	+27%	1.38
Small lagoon	+8%	0.43	+48%	1.55
Intermediate estuary	+3%	0.42	+14%	1.41
Large perennially open lagoon	+7%	0.42	+21%	1.38
Large river valley estuary	0%	0	0%	0
Fragmented river valley estuary	No Data	No Data	No Data	No Data
Open bay/harbor	No Data	No Data	No Data	No Data

Notes. Increased likelihood of high closure risk and the resultant increases in water levels when a system is presumed to be closed. Changes calculated using 2016 as the baseline. Values for change in estuary 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.

among archetypes and neither was found to be significantly different across archetypes.

3.2 | Wetland habitat change and loss

Estimates of subtidal, unvegetated mudflat, and vegetated marsh areas were made for each of the 105 sites under current, 2050 (0.6 m), and 2100 (1.7 m) projected sea levels. Site-specific estimates of wetland habitat area were aggregated to the region and to the regional archetypes for large-scale inferences into SLR response (Figure 3). Region-wide, current vegetated marsh habitats encompass 26.6 km^2 and are predicted to decrease to 19.9 km^2 with 0.6 m SLR and 8.3 km^2 with 1.7 m SLR, which represent losses of 25.3% and 68.8%, respectively. Unvegetated mudflat habitats account for 12.5 km^2 of the regional habitat under current conditions and are predicted to increase to 14.4 km^2 with 0.6 m SLR and decrease to 12.1 km^2 with 1.7 m SLR, representing a 15.7% gain and a 3.0% loss, respectively. Combined wetland habitat (vegetated marsh and unvegetated mudflat) for the entire region currently comprises 39.1 km^2, or 30.8%, of the total wetland area (Table 4). When open bays and harbors, which are predominately subtidal sites, are excluded, the current proportion of combined wetland habitat in the region is approximately 70%. With SLR, the combined wetland area across the entire region (including open bays and harbors) is predicted to decrease to 34.3 km^2 by 2050 and 20.4 km^2 by 2100 (Table 4). This represents regional losses of 12.3% and 47.8% of combined wetland habitat with 0.6 m and 1.7 m SLR, respectively.

Percent change in combined wetland habitat for the regional archetypes provides a more detailed look at SLR response (Table 4). Small archetypes with an initially low proportion of wetland habitat are predicted to experience rapid habitat loss. Specifically, small creeks currently make up 0.08 km^2 of wetland habitats in the region but are estimated to decline to 0.06 km^2 (−25%) by 2050 and to 0.01 km^2 (−88%) by 2100. Similarly, small lagoons comprise 0.08 km^2 of existing wetland areas in the region but will be reduced to 0.008 km^2 (−80%) by 2050 and only 0.001 km^2 (−98%) by 2100. Larger archetypes with higher initial proportions of wetland habitats, such as intermediate estuaries, large lagoons, and large river valley estuaries, are expected to experience slight reductions in the percent of wetland area with 0.6 m SLR, however, with 1.7 m SLR these archetypes will experience more drastic declines in wetland area, reflecting overall gains in subtidal habitat and conversion of vegetated marsh areas to unvegetated mudflats (Figure 3). For example, intermediate estuaries comprise 14.9 km^2 of existing wetland areas in the region and will lose 9.4% of wetlands by 2050 and 33.6% by 2100 (Table 4). In the course of this overall reduction in wetland area, this archetype is predicted to gain 7.8% of unvegetated mudflat areas from 2050 to 2100 at the expense of vegetated marsh (Figure 3). Wetland habitat loss, as an estimate of percent change, is predicted to be minimal in open bays and harbors, which are large, predominantly deep subtidal systems and are often highly modified and managed.

Site-specific estimates of SLR-driven change to wetland habitat areas provide insights into the range of potential responses within archetypes (Figures 2 and 4). Percent difference in wetland area per site reflects the combined loss of vegetated marsh and unvegetated mudflat compared to the current proportion of wetland area per site (Figure 2). With 0.6 m SLR, percent difference in wetland habitat per site ranged from −1.1 to 74.2%, indicating a highly variable response to SLR given site-specific model parameterization. Similarly, 1.7 m SLR estimates indicate differential site response but overall increased percent difference in wetland habitat ranging from −0.7 to 94%. Low or negative predicted percent difference in estimates suggest that some sites may be able to maintain or gain wetland habitats (vegetated marsh and unvegetated mudflat) while others will likely lose habitat area.

3.3 | Model uncertainty and sensitivity

Uncertainty analyses revealed the ranges of potential habitat outputs when input error was propagated through the model (Figure 5). The estimated range of error with 0.6 m SLR is 12.9–33.7 km^2 for vegetated marsh, 4.1–20.3 km^2 for unvegetated mudflat, and 80.0–97.2 km^2 for subtidal habitats. For 1.7 m SLR, the estimated range of error is 5.7–37.5 km^2 for vegetated marsh, 1.6–69.8 km^2 for unvegetated mudflat, and 21.2–113.8 km^2 for subtidal habitats. When the range of error was compared to the model estimates 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 (Figure 5). Although this renders model predictions for vegetated areas as conservative, the gains in vegetated areas suggested by the uncertainty analysis reflect unlikely scenarios of low SLR with high accretion. Overall, predictions for 2100 exhibit higher uncertainty due to model inputs than those for 2050.

The sensitivity analysis revealed that model estimates of wetland habitat area are most sensitive to initial elevation and SLR, followed by accretion and mouth dynamic inputs (Figure 6). When initial elevation was increased by 50 cm, an additional 53.7% of vegetated marsh was retained with 0.6 m SLR and an additional 99% was saved with 1.7 m SLR compared to the original model estimates. Increasing SLR estimates by 50% caused reductions of 22.6% and 53.5% for vegetated marsh areas for 2050 and 2100, respectively. Decreasing SLR estimates by 50% caused gains in vegetated marsh areas but is not a realistic scenario. Increasing accretion inputs resulted in additional vegetated marsh areas (+10.3%, +57.7%), while decreasing accretion resulted in additional subtidal and unvegetated mudflat areas. Percent change to habitat areas when modifying water levels associated with mouth closure were minimal compared to other model inputs but indicate slight increases to unvegetated marsh areas when this input was increased. Overall, the sensitivity of model outputs increases from 0.6 m SLR to 1.7 m SLR.

4 | DISCUSSION

Southern California exemplifies many coastal regions threatened by SLR, where efforts to understand future impacts are challenged by

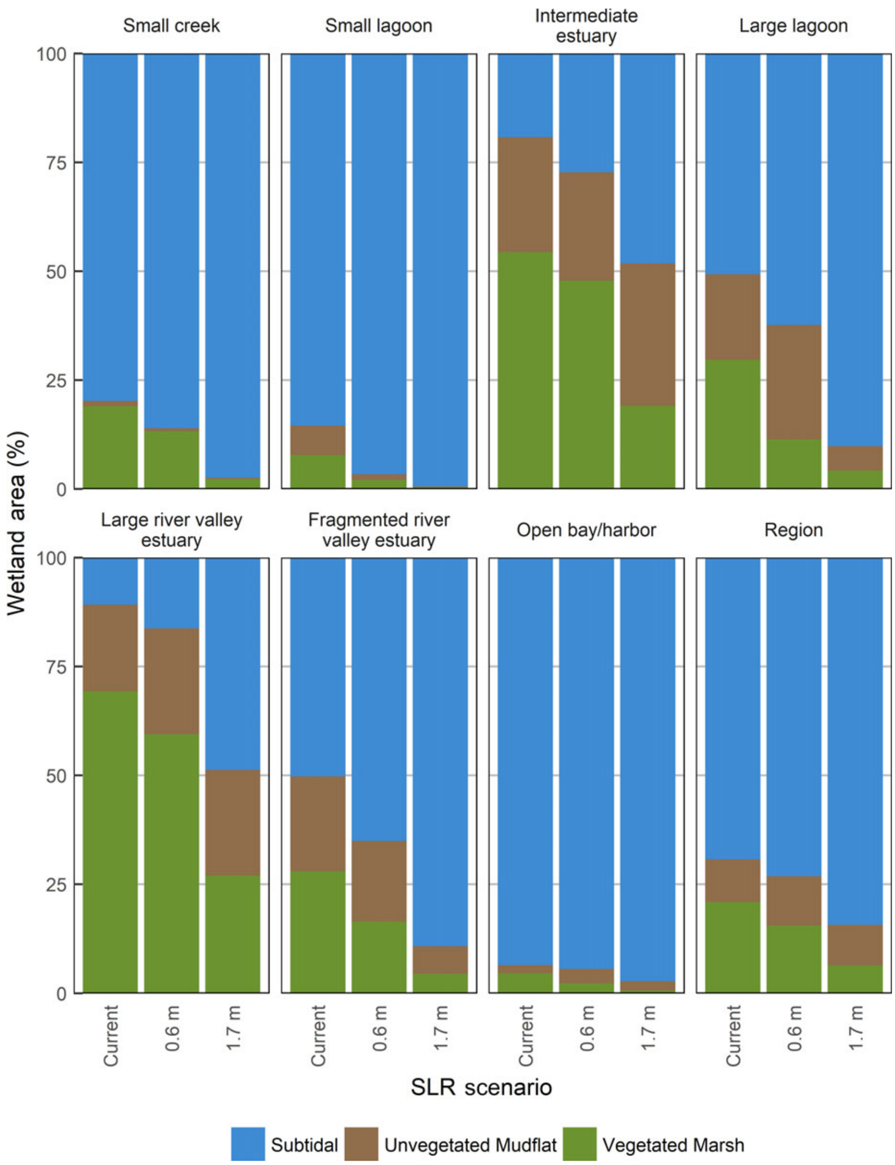

FIGURE 3 Current and predicted habitat change for wetland archetypes under two sea level rise scenarios when wetlands are confined to existing boundaries [Colour figure can be viewed at wileyonlinelibrary.com]

TABLE 4 Predicted percent change in wetland (vegetated marsh and unvegetated mudflat) habitat area for the southern California region and individual wetland archetypes

	Wetland area (km^2)			Percent change (%)	
Archetype	Existing	0.6 m SLR	1.7 m SLR	0.6 m SLR	1.7 m SLR
Small creek	0.08	0.06	0.01	25.0	87.5
Small lagoon	0.04	0.008	0.001	80.0	97.5
Intermediate estuary	14.9	13.5	9.9	9.4	33.6
Large lagoon	5.6	4.3	1.1	23.2	80.4
River valley estuary	9.4	9.2	6.2	2.1	34.0
Fragmented river valley estuary	4.1	2.9	0.9	29.3	78.0
Open bay/harbor	5.0	4.3	2.1	14.0	58.0
Region	39.1	34.3	20.4	12.3	47.8

spatial variability in drivers and responses, a heterogeneous coastal landscape, and limited data availability. Wetland hypsometry offers a standardized modeling approach that can provide multiscale insights into SLR response for a diverse region. Hypsometry data can be developed for any number of wetland sites using a consistent approach and readily available datasets. The output of hypsometric analysis provides a standardized measure of wetland elevation relative to current sea levels for each site across a large geographic area, making it a universal foundation for modeling SLR across sites of various morphologies and sizes. The majority of SLR modeling efforts are based on some metric of elevation, yet to our knowledge, wetland hypsometry as a measure of z^* has not been developed for 100+ sites at 1-m resolution to identify site-level impacts throughout a region, as we have done here. This framework can be applied to any other coastal regions where elevation data are accessible. When combined with analysis of wetland typology (or archetypes), this approach reduces the gap between fine-scale (site specific) and large-scale assessments.

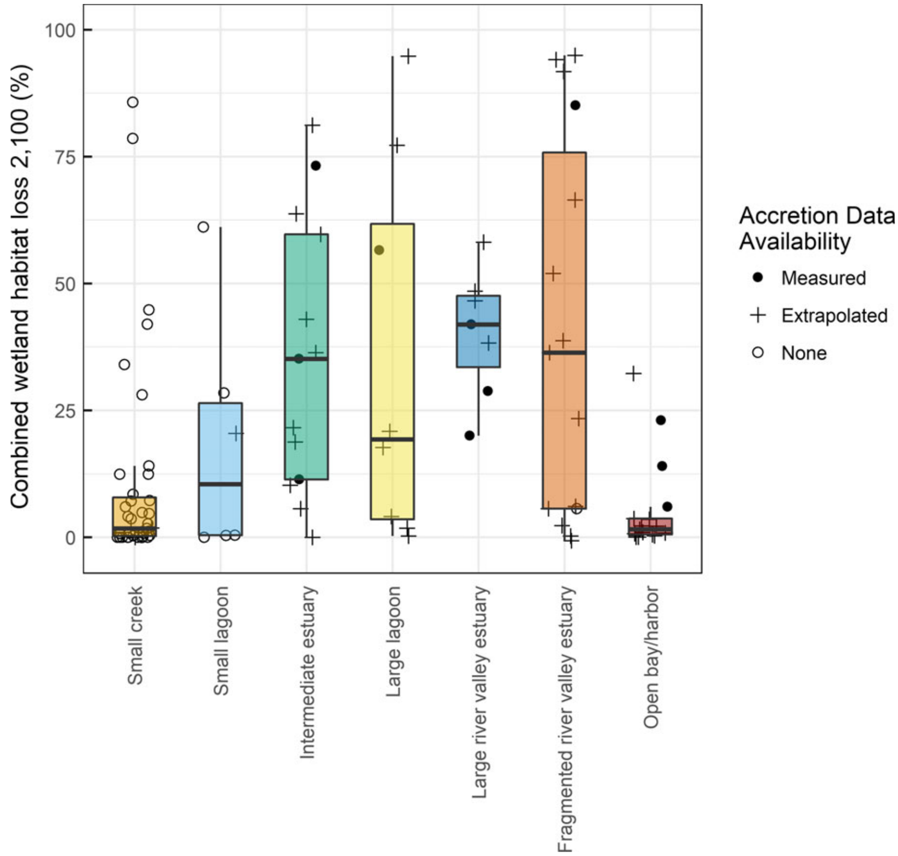

FIGURE 4 Inter- and intra-archetype variability in percent difference in combined wetland habitat (vegetated marsh and unvegetated mudflat) predicted with 1.7 m SLR by 2100. Boxplots indicate the distribution of percent difference for each archetype including the median, first, and third quartiles. Points indicate each site within the archetype class and the availability of accretion data for that site: Data were either available from our literature review (measured, closed circle), extrapolated using the archetype framework (extrapolated, plus) or unavailable (none, open circle). For sites where accretion data were unavailable (small creeks and lagoons), estimates were made using a bathtub model assuming no accretion [Colour figure can be viewed at wileyonlinelibrary.com]

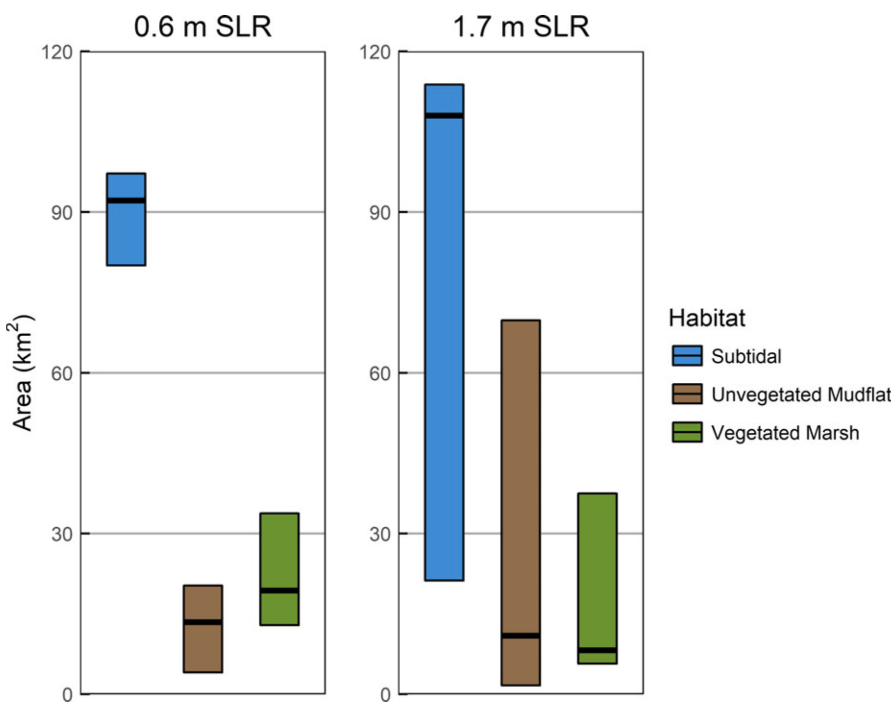

FIGURE 5 Bookended range of the potential area for each wetland habitat when error is propagated through the model. Black lines represent the area of each habitat originally predicted by the model. Floating bars represent the range of possible values when model input errors are considered

Large-scale assessments often provide a limited understanding of regional SLR response (Fagherazzi et al., 2012). Recent global assessments report wetland losses up to 22%–59% under low SLR (0.3 and 0.5 m) and 78% with high SLR (1.1 m) (Nicholls, 2004; Nicholls, Hoozemans, & Marchand, 1999; Spencer et al., 2016). Such assessments often indicate no or low vulnerability to SLR in southern California relative to other regions because these wetlands are either not included in global databases (e.g., RAMSAR) or are characterized under a broad coastal classification scheme (Nicholls et al., 1999; Spencer et al., 2016). Our findings indicate that when coastal

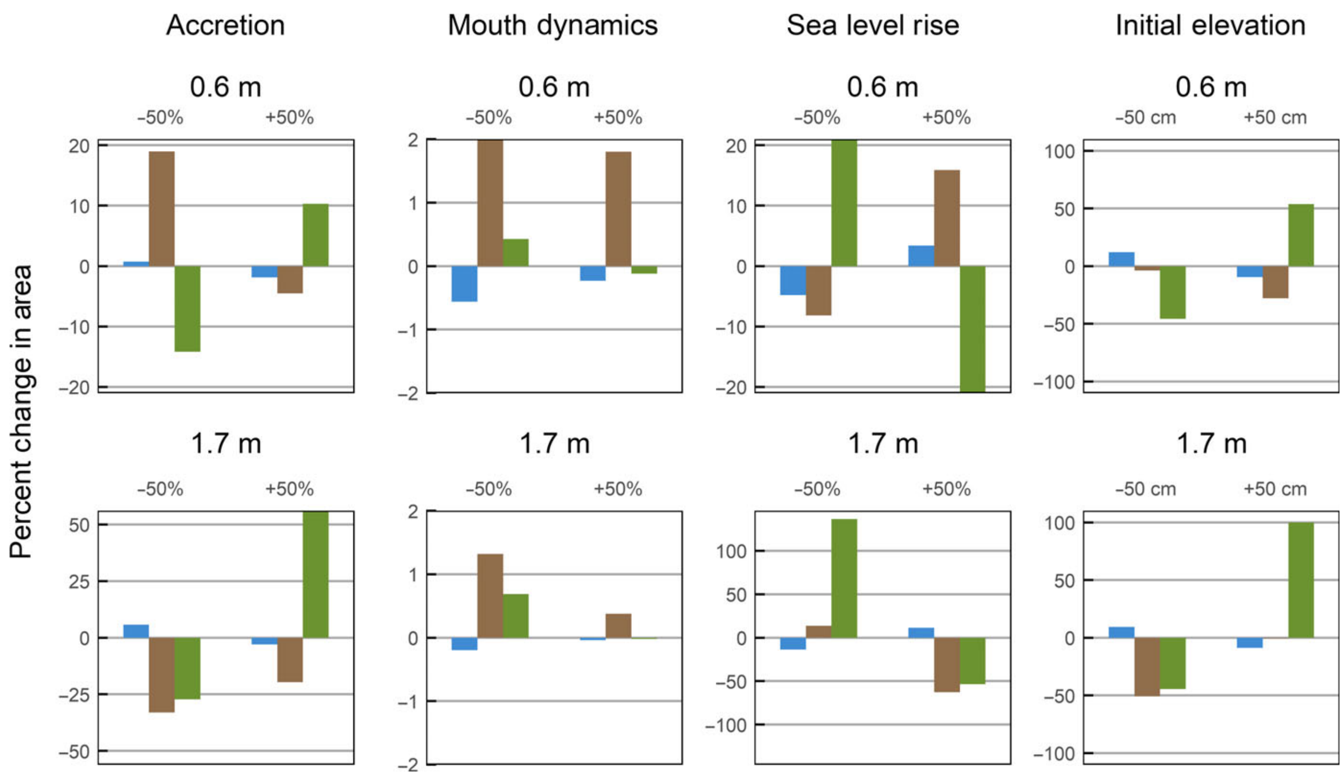

FIGURE 6 Percent change in habitat area caused by sensitivity to model inputs. Percent change is calculated based on the area of each habitat (subtidal (light gray), unvegetated mudflat (gray), and vegetated marsh (dark gray)) originally predicted by the model. Sensitivity was analyzed by varying model parameters: accretion, mouth dynamics, SLR by ±50%, and initial elevation by ±50 cm. Note varying *y*-axis scales in plot panels [Colour figure can be viewed at wileyonlinelibrary.com]

wetlands are confined to their existing extents (i.e., not allowed to migrate), the southern California region could experience combined wetland habitat losses of 12% and 48% with 0.6 m and 1.7 m of SLR. For vegetated marsh alone, these figures increase to 25% and 68% loss, respectively. These projected losses fall within the range of global assessments and are comparable to similar regional assessments (e.g., 45% loss with 0.69 m SLR in Georgia (Craft et al., 2009)). Our modeling approach provides reasonable SLR response estimates at large spatial scales, but also facilitates assessment at smaller scales to provide contextualized intraregional patterns in SLR response.

Wetland SLR response at intermediate scales is a critical missing piece in predicting SLR impacts globally. To address this, our assessment characterized variability in SLR-driven habitat loss among and within coastal wetland archetypes. Through the use of archetypes in southern California, we were able to identify wetland types that have a higher risk of habitat loss and to elucidate the underlying processes that lead to habitat loss. Many of the factors contributing to high SLR vulnerability in this study region are universal and applicable worldwide. For example, large bays and harbors are predicted to experience minimal relative loss of vegetated marsh and unvegetated mudflat habitats because existing habitat composition in these systems is predominantly subtidal with few fringing wetlands. Also, bays and harbors are often armored and highly managed to maintain deep water and remain open to the ocean; consequently, most impacts will be to subtidal habitats. Conversely, small creeks and lagoons are at greater risk of more rapid loss of wetland because of their small extent and the steep grade of adjacent upland transition zones, which limits migration potential (Donnelly & Bertness, 2001; FitzGerald, Fenster, Argow, & Buynevich, 2008). Moreover, these small systems often have limited opportunity for accretion, decreasing their ability to accommodate SLR.

Insights at the archetype level are especially valuable for intermediate estuaries, which exhibit a wide range of potential responses to SLR. This archetype is characteristically diverse because it consists of systems that are naturally dynamic, fluctuating between open and closed mouth states due to fluvial, tidal, and wave drivers (Behrens et al., 2015). These dynamic estuarine systems are present along the coasts of the Western United States, Mexico, South America, Europe, South Africa, Asia and Australia (McSweeney, Kennedy, Rutherfurd, & Stout, 2017) and are often underrepresent in large-scale assessments despite offering important ecosystems services such as refuge for endangered fish species (e.g., Swenson, 1999). Incorporating estuary mouth dynamics into SLR response models is challenging because mouth dynamics are expected to be heavily influenced by SLR (Haines & Thom, 2007) and are difficult to model. Therefore, the difficulty in predicting mouth dynamics means that the intermediate estuary response to SLR also comes with a high amount of uncertainty. Given that intermediate estuaries are more widespread globally than previously reported (McSweeney et al., 2017),

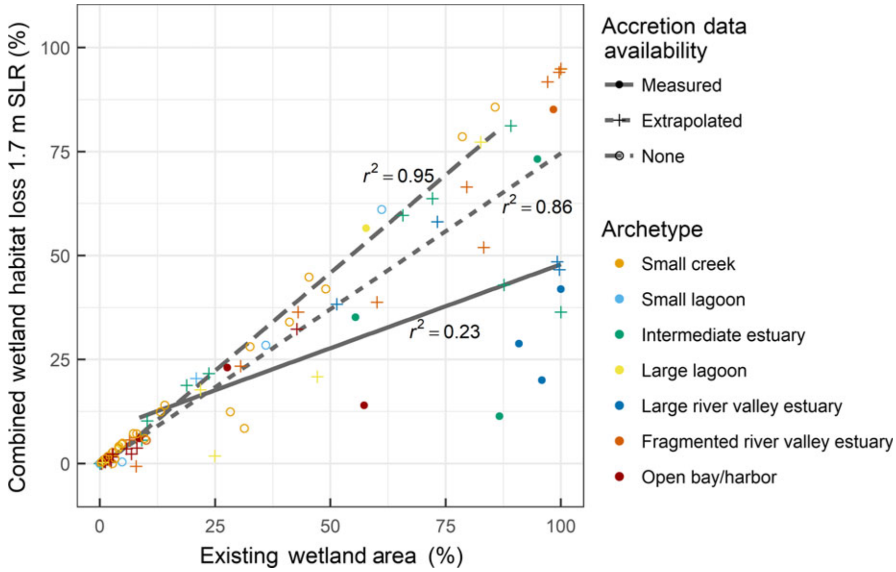

FIGURE 7 Relationship between predicted percent difference in wetland habitat with 1.7 m SLR by 2100 and existing (2016) percent wetland areas (vegetated marsh and unvegetated mudflat) for all 105 sites. Site symbology reflects the archetype classification (color) and data availability of accretion inputs for that site (shape). Accretion data were either available from our literature review (closed circle, solid trendline), extrapolated using the archetype framework (plus, dashed trendline) or unavailable (open circle, long-dash trendline). For sites where accretion data were unavailable (small creeks and lagoons), estimates were made using a bathtub model assuming no accretion [Colour figure can be viewed at wileyonlinelibrary.com]

additional monitoring and modeling of estuary dynamics relative to coastal and fluvial processes also subject to climate change represent a high research priority.

Similarly, fragmented river valleys exhibit the largest range of SLR responses because they also encompass a diverse group of wetlands, all of which have been highly modified by humans. This archetype exemplifies the global phenomenon of coastal squeeze caused by shoreline hardening, where natural systems have become hydrologically altered and bounded by concrete structures such as seawalls and jetties (Pontee, 2013). Anthropogenic impacts such as habitat fragmentation and shoreline hardening in coastal wetlands will accelerate habitat loss with future SLR (Gittman et al., 2015). However, even archetypes considered to be unmodified are also highly vulnerable to SLR. Larger archetypes, including intermediate estuaries, large lagoons, and river valley estuaries, will experience the greatest amount of habitat change in the region. Although these systems are characteristically large, predominantly comprised of wetland habitat, and have shallow grades ideal for upland migration, it is likely that allochthonous and autochthonous inputs to accretion will not be sufficient to maintain elevation with SLR in the long term. Within these larger archetypes, there is also wider range of site responses, which reflects the intra-archetype diversity of system characteristics.

Indeed, we found that site-level response to SLR is highly variable, further emphasizing the importance of spatial scale in SLR assessments and the need to supplement regional projections with local insights. To date, local insights are best acquired through fine-scale SLR assessments conducted for individual wetland sites. This is exemplified by the work of Thorne et al. (2016, 2018) for several wetland sites in southern California including Upper Newport Bay, Mugu Lagoon, and Tijuana River Estuary. Detailed field collections of physical and biological data were used to parameterize the process-based WARMER model (Swanson et al., 2014) to predict SLR response for a subset area within the study sites. Our SLR response predictions corroborate their overall finding that these sites will experience conversion of mid and high marsh to low marsh with modest SLR projections (0.44 m) and gains of intertidal mudflat and subtidal habitat at the expense of vegetated marsh with high SLR projections (1.66 m) (Thorne et al., 2016). Specifically, projections for Upper Newport Bay by Thorne et al. (2016) indicate that 60% of the wetland area will be converted to subtidal with 1.66 m SLR, and this study estimates a total of 57% subtidal under the same SLR scenario. Mugu Lagoon is estimated to become 100% intertidal mudflat with 1.66 m SLR (Thorne et al., 2016), whereas we predict the future habitat composition to be 50% intertidal mudflat and 25% subtidal. However, our Mugu wetland boundary is larger than the area studied by Thorne et al. (2016) and includes more subtidal areas. Thorne et al. (2016) predict that a portion of the Tijuana River Estuary will convert to 80% mudflat and 20% subtidal, and we predict that a larger area of the estuary containing a higher initial proportion of vegetated marsh will consist of 32% mudflat and 24% subtidal with high SLR. Similarities in SLR response predictions for these three sites indicate that our approach may provide an alternative option when time and resource intensive assessments are not possible for individual wetlands or when regional efforts need to identify sites where more work should be targeted.

Accommodating all wetlands types regardless of data availability is another essential step toward a comprehensive assessment of wetland response. Our approach is designed for flexibility in model parameterization by incorporating site-specific data when available and by using archetype or regional defaults when site data are not available. Having predicted site-level response estimates for each site revealed additional patterns in SLR response, including a positive correlation between the initial proportion of wetland habitat in each site and the potential relative wetland habitat loss, as well as the implications of data availability (Figure 7). The availability of in situ accretion measurements determines the relationship between wetland habitat composition and wetland loss, where sites with higher estimates of accretion supported by empirical data appear to be less at risk for future habitat loss. Our model is sensitive to the accretion parameter, and our literature review of regional accretion rates highlights that more work is needed to characterize accretion variability across marsh zones and among archetypes.

Like many SLR impact assessments, a lack of data adds to the uncertainty of our estimates of SLR-driven habitat change. Our results indicate that efforts should be targeted toward providing additional, spatially explicit estimates of accretion, perhaps supplied by a coordinated regional or global sampling network (Osland et al., 2017; Webb et al., 2013). Such information would improve data gaps involving plant-mediated biogeomorphic feedbacks, especially considering that these processes are variable between individual wetlands and archetypes (Webb et al., 2013) and serve as broad indicators of vulnerability (Ganju et al., 2017). Although data limitations made it difficult to detect significant differences in accretion between archetypes, the archetype framework represents meaningful physical and ecological differences that will likely become more significant in the future with the support of additional empirical data. The mouth dynamics model component also represents an important, yet highly uncertain, hydrodynamic aspect of modeling SLR response (Rodríguez, Saco, Sandi, Saintilan, & Riccardi, 2017). As evidenced by the previous lack of understanding on the distribution of intermediate estuaries (McSweeney et al., 2017) and the complexity of modeling such systems, data availability represents a major hurdle to modeling SLR impacts in certain wetland types and regions worldwide. Furthermore, advances in SLR projections and elevation datasets that reduce uncertainty and error would be beneficial to modeling SLR response, given the high model sensitivity to SLR rate and starting elevation.

This assessment is designed to inform comprehensive coastal planning by providing insights at a variety of spatial scales to aid both regional efforts and local management. Consistent, large-scale methods are critical for informing regional planning and prioritization, whereas detailed methods can then be used to help inform site-specific designs (Runting, Wilson, & Rhodes, 2013). Our approach augments existing SLR response assessments by estimating both the risk to individual wetlands and the relative SLR response within a region. Our findings are comparable to both large-scale assessments and intensive, site-based models, making it a useful tool for coastal management. Multiscale inferences to SLR impacts can also aid coordinated regional efforts in maintaining regional wetland composition by indicating which subregions, archetypes, or individual sites have a higher relative vulnerability to SLR (Stralberg et al., 2011). Failure to include all relevant coastal wetland types can produce an incomplete picture of regional vulnerability and can obscure differences in archetype response to SLR. This is exemplified by intermediate estuaries which are globally important wetland types and require additional management considerations when planning for SLR impacts. Our approach can also be used to investigate how alternative management strategies could reduce SLR-driven habitat loss by manipulating model inputs (see SCWRP, 2018 for more details). For example, expanding hypsometric curves to include suitable upland habitat adjacent to a given site can reveal reductions in habitat loss associated with facilitating wetland migration. Raising the starting elevation of hypsometric curves could indicate the potential of thin-layer sediment augmentation as an adaptation strategy. Furthermore, modifying accretion rates can help set sediment management goals required to maintain current wetland habitat composition within a site or across the region. Overall, this approach is suitable for management applications in other regions given its ability to accommodate a diversity of wetland types and varying levels of data availability.

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ORCID

Cheryl L. Doughty http://orcid.org/0000-0003-3802-9813

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of the article.

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