Articles

Tidal Marsh Susceptibility to Sea-Level Rise: Importance of Local-Scale Models

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Abstract

Increasing concern over sea-level rise impacts to coastal tidal marsh ecosystems has led to modeling efforts to anticipate outcomes for resource management decision making. Few studies on the Pacific coast of North America have modeled sea-level rise marsh susceptibility at a scale relevant to local wildlife populations and plant communities. Here, we use a novel approach in developing an empirical sea-level rise ecological response model that can be applied to key management questions. Calculated elevation change over 13 y for a 324-ha portion of San Pablo Bay National Wildlife Refuge, California, USA, was used to represent local accretion and subsidence processes. Next, we coupled detailed plant community and elevation surveys with measured rates of inundation frequency to model marsh state changes to 2100. By grouping plant communities into low, mid, and high marsh habitats, we were able to assess wildlife species vulnerability and to better understand outcomes for habitat resiliency. Starting study-site conditions were comprised of 78% (253-ha) high marsh, 7% (30-ha) mid marsh, and 4% (18-ha) low marsh habitats, dominated by pickleweed *Sarcocornia pacifica* and cordgrass *Spartina* spp. Only under the low sea-level rise scenario (44 cm by 2100) did our models show persistence of some marsh habitats to 2100, with the area dominated by low marsh habitats. Under mid (93 cm by 2100) and high sea-level rise scenarios (166 cm by 2100), most mid and high marsh habitat was lost by 2070, with only 15% (65 ha) remaining, and a complete loss of these habitats by 2080. Low marsh habitat increased temporarily under all three sea-level rise scenarios, with the peak (286 ha) in 2070, adding habitat for the endemic endangered California Ridgway's rail *Rallus obsoletus obsoletus*. Under mid and high sea-level rise scenarios, an almost complete conversion to mudflat occurred, with most of the area below mean sea level. Our modeling assumed no marsh migration upslope due to human levee and infrastructure preventing these types of processes. Other modeling efforts done for this area have projected marsh persistence to 2100, but our modeling effort with site-specific datasets allowed us to model at a finer resolution with much higher local confidence, resulting in different results for management. Our results suggest that projected sea-level rise will have significant impacts on marsh plant communities and obligate wildlife, including those already under federal and state protection. Comprehensive modeling as done here improves the potential to implement adaptive management strategies and prevent marsh habitat and wildlife loss in the future.

Keywords: climate change; endangered species; marsh; sea-level rise; resilience; San Francisco Bay; wetland

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Introduction

Coastal ecosystems are vulnerable to climate change with increased flooding from sea-level rise and accompanying changes in storm frequency and intensity that may increase flooding and coastal erosion (IPCC 2007; Kirwan and Murray 2007; Solomon et al. 2009). Changes in ocean temperature, local freshwater delivery, and ocean acidification also have potential negative impacts on these systems (IPCC 2007; Fitzgerald et al. 2008; Nicholls and Cazenave 2010). Projections of sea-level rise are dependent on carbon dioxide emissions; ocean thermal expansion; and melting of land-based ice from glaciers, ice caps, and ice sheets (IPCC 2007; National Research Council 2012). Current projections of global sea-level rise range from 57–110 cm (Jevrejeva et al. 2012) to 75–190 cm (Vermeer and Rahmstorf 2009) by 2100, with acceleration in the later part of the century. For the Pacific coast of North America, recent sea-level rise projections range from 42 to 167 cm by 2100 (National Research Council 2012). The expected accelerated rate of sea-level rise through the 21st century will put many coastal ecosystems and the species that depend on them at risk (Thorne et al. 2012; Woodrey et al. 2012).

Tidal marshes are transitional ecosystems between land and sea and are found along low-energy intertidal coastlines. They are influenced by regular flushing from tidal action and storms (Mitsch and Gosselink 1993). These marshes are highly productive and are dominated by halophytic plants that have varying tolerances to tidal inundation and salinity, leading to zonation across the elevation tidal gradient (Mancera et al. 2005; Schile et al. 2011). Decomposition and accumulation of below- and above-ground organic matter, combined with surface mineral deposition, can allow marshes to maintain their elevation relative to local sea levels given enough time (Morris et al. 2002; Gedan et al. 2011). However, a decrease in elevation relative to sea level may occur if inundation frequency outpaces vertical accretion (Reed 2002). This elevation decrease may result from limited sediment supplies (Blum and Roberts 2009) and degradation of local natural biogeomorphic processes (Bouma et al. 2008), both of which have been shown to be important for marsh stability (Cahoon and Reed 1995). The removal or dieback of vegetation may cause soil compaction by root collapse and erosion of the surface, thereby accelerating marsh drowning with sea-level rise (Cahoon 2006; Day et al. 2011).

Coastal estuaries in many regions worldwide have been fragmented and modified to an extent that natural processes are limited, consequently degrading wetland response to relative sea-level rise. Damming and intensive human activities such as levee construction and water diversion can reduce sediment delivery rates to estuaries and can decrease marsh accretion potential or accelerate erosion (Ma et al. 2014). Coastal marshes have been negatively impacted by human land-use change and habitat degradation (Nichols et al. 1986; Takekawa et al. 2006; McGranahan et al. 2007; Brusati and Grosholz 2009), with approximately 52% of the U.S. population living in coastal watershed counties (NOAA's State of the Coast 2013). Marshes support a relatively low diversity of wildlife species; however, many are either endemic or habitat specific or are geographically restricted subspecies (Greenberg et al. 2006). Due to indirect human impacts, many marsh wildlife species are listed as endangered or threatened pursuant to the US Endangered Species Act (ESA 1973, as amended). Examples from the Atlantic and Pacific coasts of North America include the Florida salt marsh vole *Microtus pennsylvanicus dukecampbelli*, Belding's savannah sparrow *Passerculus sandwichensis beldingi*, Lower Keys marsh rabbit *Sylvilagus palustris hefneri*, and maritime ringlet butterfly *Coenonympha nipisiquit*, all of which have decreased substantially in numbers as a result of habitat loss (Woods 1992; U.S. Fish and Wildlife Service [USFWS] 1991, 2007).

Marshes and the wildlife populations they support are particularly vulnerable to sea-level rise, where loss of habitats and variations in water depth and duration play major roles in structuring these plant and wildlife communities (Brittain and Craft 2012). When assessing ecosystem impacts, the scope and scale of quantitative assessments for sea-level rise should be at an ecologically relevant scale for the organisms. If not, then predicting wildlife species response is difficult, as marsh habitats are locally regulated by wetland processes, including tidal inundation (Woodrey et al. 2012). The tolerance to disturbance, including inundation, may vary significantly between species and contributes to state changes in community composition and structure. Plant community–based wildlife habitat relationships (Table 1) can be used as the first step in deriving future species distribution models (Guisan and Thuiller 2005; Veloz et al. 2012) that can be related and used to assess climate change impacts (Seavey et al. 2011; Traill et al. 2011) and to develop climate change adaptation strategies by managers.

In the San Francisco Bay estuary, California, USA, we evaluated the susceptibility of tidal marsh habitats and their wildlife species to sea-level rise by using an empirical marsh response model built with site-specific data. We aimed to assess marsh response to sea-level rise by coupling site-specific datasets for elevation, local inundation, and accretion with plant community–based wildlife habitat relationships. We derived plant community distribution models, grouped as low, mid, and high marsh habitats, to model wildlife habitat state changes to 2100 under three sea-level rise scenarios (National Research Council 2012). Lastly, we compared our modeling approach and outputs to those of other modeling efforts for this area to evaluate the relevance of these modeling efforts to management decision making.

Methods

Study area

San Francisco Bay is the largest estuary on the Pacific coast of North America and is an important site for

Table 1. Representative tidal marsh wildlife species and their common plant community relationships were used to understand how changes in habitats with sea-level rise could impact wildlife for San Pablo Bay National Wildlife Refuge. Many species will move between plant community types for foraging and nesting and will seek refuge in uplands during high tidal inundation. Plant communities were defined from 2008 field surveys and related to elevations relative to mean sea level (MSL).

Family name	Scientific name	Common name	Community
Parulildae	*Geothlypis trichas sinuosa*	Salt marsh common yellowthroat[a]	High-upland
Emberizidae	*Melospiza melodia samuelis*	San Pablo song sparrow[a]	High-upland
Rallidae	*Laterallus jamaicensis coturniculus*	California black rail[b]	Mid-high
Muridae	*Mus musculus*	House mouse[c]	Mid-high
Cricetidae	*Microtus californicus sanpabloensis*	San Pablo vole[a]	Mid-high
Cricetidae	*Reithrodontomys raviventris*	Salt marsh harvest mouse[d]	Mid-high-upland
Accipitridae	*Circus cyaneus*	Northern harrier[c]	Low-high-upland
Accipitridae	*Elanus leucurus*	White-tailed kite[c]	Low-high-upland
Rallidae	*Rallus limicola*	Virginia rail[c]	Low-high
Rallidae	*Porzana Carolina*	Sora[c]	Low-high
Troglodytidae	*Cistothorus palustris*	Marsh wren[c]	Low-high
Rallidae	*Rallus obsoletus obsoletus*	California Ridgway's rail[d]	Low

[a] State species of special concern (California Department of Fish and Wildlife 2015b).
[b] State threatened (California Department of Fish and Wildlife 2015a).
[c] No status.
[d] Federal and state endangered (California Department of Fish and Wildlife 2015a; USFWS 2015).

migratory birds and endemic wildlife (Takekawa et al. 2006). The tidal regime is mixed semidiurnal, with an average diurnal range of 1.78 m (Golden Gate tide gauge, National Oceanic and Atmospheric Administration; station 9414290, 37°48.4′N, 122°27.9′W 37°; Golden Gate hereafter). Sea level has risen 19.3 cm between 1900 and 2000 in San Francisco Bay (Cayan et al. 2006), with future projections of an increase of up to 1.67 m by 2100 (National Research Council 2012). Our study examined a 324-ha high marsh platform in the western portion of the USFWS San Pablo Bay National Wildlife Refuge, hereafter SPBNWR (38°08′N, 122°24′W), located along the northern edge of the San Pablo Bay, a subembayment in the northwest reach of the San Francisco Bay estuary (Figure 1). The landward edge of SPBNWR is bounded by levees that protect private agriculture and pasturelands and a state highway. The refuge is comprised of extensive tidal marsh, restored wetland areas, and shallow mudflat habitat, providing winter and year-round habitat for thousands of migratory waterbirds and resident marsh wildlife (Takekawa et al. 2006; Jaffe et al. 2007). Pickleweed *Sarcocornia pacifica* is the predominant marsh vegetation type, with a small baywide strip of cordgrass *Spartina* spp. San Pablo Bay tidal marshes have been greatly impacted, with nearly an 85% loss from human activities such as diking, mining, salt pond development, road construction, and farming (USFWS 2007).

The SPBNWR is home to endemic and specialized species, including those that receive state and federal conservation status (ESA 1973; Table 1), such as the salt marsh harvest mouse *Reithrodontomys raviventris* (California Department of Fish and Wildlife 2015a; USFWS 2015) and California black rail *Laterallus jamaicensis coturniculus* (California Department of Fish and Wildlife 2015a), both of which require pickleweed for cover and nesting (Hulst 2000; Tsao et al. 2009). The endangered endemic California Ridgway's rail *Rallus obsoletus obsoletus* (ESA 1973) also resides at SPBNWR and uses cordgrass for cover and nesting (USFWS 1984). Marsh wildlife with limited dispersal ability often are forced to move out of their protective habitats to adjacent marsh-upland areas during high flood events, thereby exposing them to predators, competition, and drowning (Evens and Page 1986; Zedler 2010).

Marsh elevation

To assess the topography of SPBNWR, two comparable high-resolution elevation surveys were done 13 y apart across the 324-ha study site, a western portion of SPBNWR (see Thorne et al. 2014). The first survey-grade elevation survey (orthometric heights ±2.0 cm) was performed in 1995 by establishing 15 horizontal and vertical control stations. In total, six benchmarks were used to provide North American Vertical Datum of 1988 (NAVD88) vertical control. Using standard leveling techniques, transects were run from and between the 15 control stations approximately 50–100 m apart. The second survey-grade elevation surveys were performed in 2008 with a Leica RX1200 real-time kinematic global positioning system (GPS) rover (x,y accuracy ±1 cm, orthometric heights accuracy ±2 cm; Leica Geosystems Inc., Norcross, GA). Rover positions were received from the Leica Smartnet system (www.leica-geosystems.com) and referenced to a National Geodetic Survey benchmark (X 552 1956, Mare Island). The average measured vertical error for the benchmark was ±2.0 cm. Elevation was surveyed perpendicular to the bay edge, with a survey point taken every 25 m; 50 m separated transect lines. For both models, the Geoid03 model was used

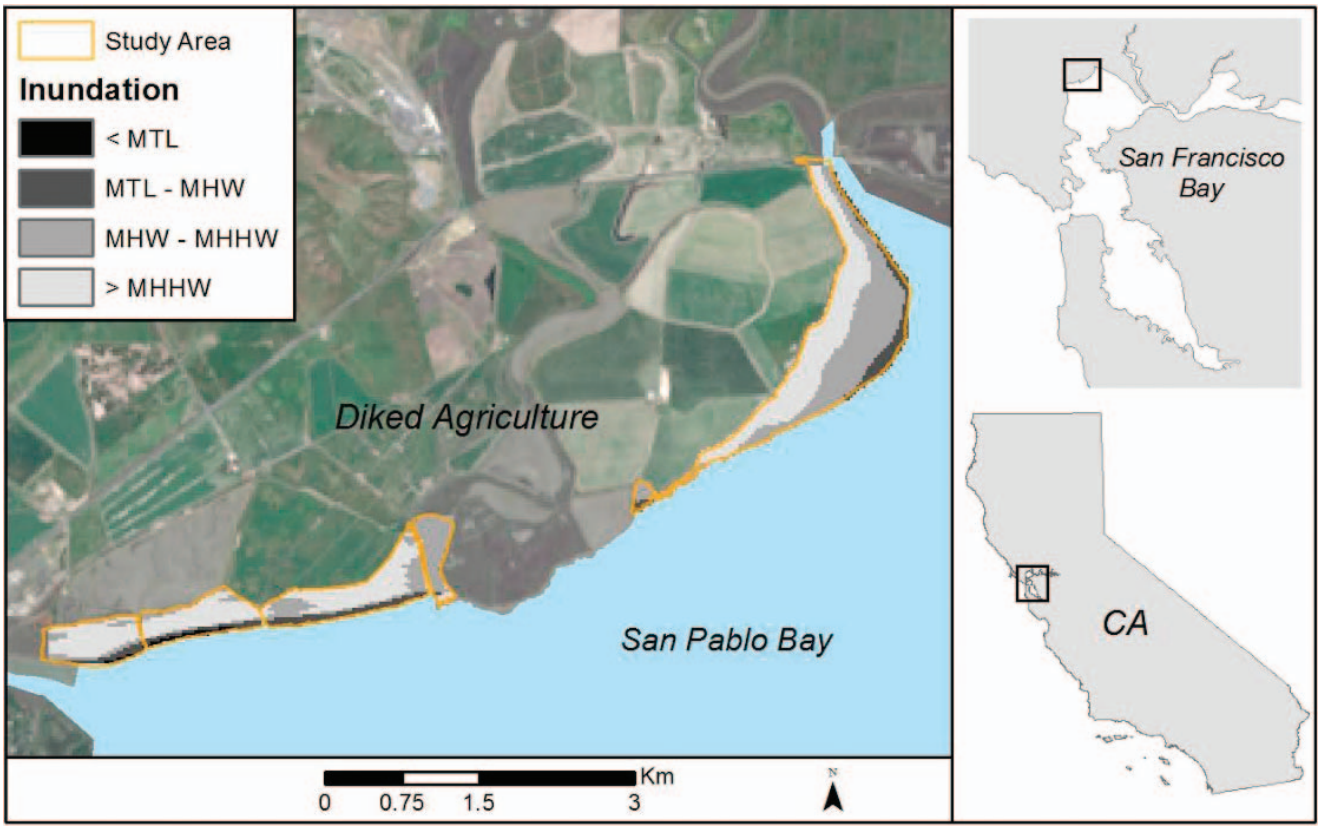

Figure 1. San Pablo Bay National Wildlife Refuge, located in the northern reach of San Francisco Bay, California, USA, includes 324 ha of tidal marsh habitat that was modeled for projected sea-level rise and plant community response in 2008. It is adjacent to leveed agricultural lands and a large shallow bay to the south.

in calculating elevations from orthometric heights (NAVD88), and all points were projected to North American Datum of 1983 (NAD83) Universal Transverse Mercator Zone 10.

These two elevation datasets were synthesized to create digital elevation models (DEMs), an elevation raster in ArcGIS 9.3 Spatial Analyst (ArcGIS Desktop Release 10.1; Environmental Systems Research Institute, Redlands, CA) with Kriging methods (30 $\times$ 30 m cell size). The exponential model for Ordinary Kriging was used, and model parameters were adjusted to minimize the root mean square error (RMSE), an internal measure of model performance. Lag size and number of lags were optimized for the site (lag size $\times$ lag number $<$ one-half maximum distance among points); an anisotropy adjustment was applied because of a trend in the elevation data. Resultant models were cross-validated by comparing accuracy of predictions from models created with 70% of the data against the remaining 30% of the data.

Marsh elevation change was estimated by comparing two high-resolution elevation surveys done 13 y apart and was used as a proxy for accretion or subsidence (see Thorne 2012; Thorne et al. 2014). Positive values indicate an increase in elevation (i.e., accretion) and negative values indicate a decrease in elevation (i.e., subsidence) from 1995 to 2008. By calculating elevation differences using 2008 and 1995 raster DEMs, we assumed that this would account for any surface or subsurface processes occurring, such as organic matter deposition, decomposition, sediment accumulation, and compaction (Cahoon et al. 1996).

Vegetation survey

At 511 elevation points, plant species, height (mean, maximum, measured within 0.05 m), and percent cover were recorded within 0.25-m^2 quadrats (see Thorne 2012; Thorne et al. 2014). For all plant species within a 0.25-m^2 quadrat, average and maximum height (measured to the nearest 1 cm) were measured visually, along with estimated absolute percent cover. Average height was obtained by visually assessing the most dominant canopy height for each species and measuring a single plant within that canopy. Vegetation presence data were used to define elevation zones relative to mean sea level (MSL, in meters) and which plants occurred to define low, mid, and high marsh, or marsh-upland transition zones. Intertidal mudflat comprised the zone between mean lower low water and the lowest measured extent of tidal marsh vegetation (typically roughly at mean tide level). We used these vegetation and elevation relationships to predict transitions or state changes of wildlife habitats to assess vulnerability and to better understand outcomes for habitat persistence to 2100 for management.

Inundation data

Water level data loggers (model 3001, 0.01% full scale resolution; Solinst Canada Ltd., Georgetown, Ontario, Canada) were deployed at four locations within SPBNWR. They were placed at the mouth and upper reaches of two second-order channels (tidal creeks) to capture the local tidal cycle and inundation patterns for different habitat areas. We collected continuous data every 6 min throughout 2010 to develop inundation rates. Loggers were surveyed with the real-time kinematic GPS at the time of deployment and at each data download to correct for any movement. Water levels were corrected for local barometric pressure, with data from independent barometric loggers deployed at the study site (model 3001, 0.05% full scale accuracy; Solinst Canada Ltd.). Mean high water (MHW) and mean higher high water (MHHW) tidal datums were calculated by averaging the four water level logger 2010 high water peaks.

The relationship between water levels at the Golden Gate tide gauges and SPBNWR water level loggers was used to relate inundation levels at the SPBNWR site and so relate these levels with measured accretion for modeling input. This relationship was determined by dividing tide height for a given tide at SBPNWR by those at the Golden Gate and was used to develop a SPBNWR conversion factor (C), based on the Golden Gate high tide height (h), and was estimated using a quadratic equation ($r^2 = 0.36$):

$$C = 0.20h^2 - 0.91h + 2.01 \quad (1)$$

Multiplying C by the Golden Gate tide height provided hindcasted values for inundation for 1995–2008. We then used the hindcasted water data to determine annual inundation depth, duration, and frequency at each 2008 real-time kinematic survey location in SPBNWR. We validated our predictions by first calculating the RMSE from comparing hindcasted 2010 Golden Gate data to observed 2010 SPBNWR peak tide heights (RMSE = 0.04 m). Next, we compared the RMSE between observed Golden Gate data to SPBNWR peak tide heights. Given the importance of water height above the marsh plain for suspended sediment settling and accretion rates, we used MHHW levels to project sea-level rise to 2100 due to the high marsh platform at this site.

Model development

We developed a generalized additive model (GAM) for elevation change by using observed changes in elevation to drive model response to changes in MHHW and sea-level rise. We used the R package mgcv version 1.8-4 to process the GAM model (http://cran.r-project.org/web/packages/mgcv/index.html). Using Golden Gate gauge data from 1980 to 2008, we determined the relationship between annual MHHW and the flooding characteristics inundation frequency, inundation depth, and flooding duration across a marsh (0.5–2.3 m NAVD88, 0.1-m intervals). The yearly summed inundation depths were found to be most positively correlated with MHHW ($r^2 = 0.96$); therefore, water projections related to MHHW were used when modeling sea-level rise scenarios. The flooding characteristics frequency, depth, and duration were highly correlated (> 0.7); thus, we selected inundation depth as the flooding characteristic to use. The GAM fits inundation depth (independent variable) to elevation change (dependent variable) by using regression splines (Figure 2).

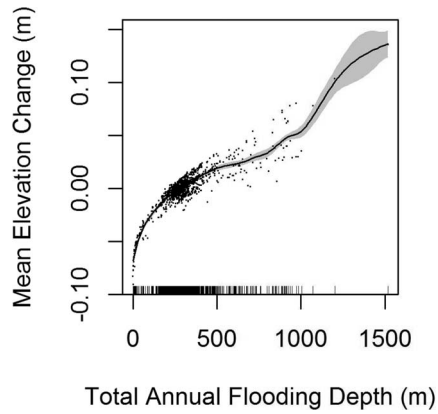

Figure 2. Relationship between total annual flooding depth (m) and mean annual elevation change (m) from 1995 to 2008 at San Pablo Bay National Wildlife Refuge, located in the northern reach of San Francisco Bay, California, USA, by using regression splines in a generalized additive model. This relationship between tidal marsh inundation and changes in elevations was used as an input to model future elevations.

We used a raster grid cell–based model (30 × 30 m) to examine the spatial response of SPBNWR marsh surface with sea-level rise. This marsh response model accounted for vertical accretion and subsidence within each model grid cell. Our model assumed that the change in marsh elevation (E) in a given grid cell from year t to the next year $t + 1$ is a function of annual inundation depth (D_t), which is a function of initial elevation (E_t), and MHHW (H_t).

$$E_{t+1} = D_t \quad (2)$$

$$D_t = E_t + H_t \quad (3)$$

We used the hindcasted water level data to determine yearly inundation depth, frequency, and duration at every 2008 elevation point. The GAM with only inundation depth had an $r^2 = 0.87$; thus, we used it to project future elevation change under sea-level rise scenarios. Distance to bay edge was considered an independent variable due to its importance in previous analysis (Thorne et al. 2014); however, because it only marginally improved model fit ($r^2 = 0.88$), we excluded it in the final analyses to simplify model implementation. For each cell, D_t, defined by equations (2) and (3), was used to determine elevation change from projections with the GAM. The standard error of the GAM result was used to determine the range around the mean, thereby accounting for one source of model uncertainty in the final results.

The model was optimized by adding a coefficient to the predicted amount of elevation change and by

running the 'optim' function in R (www.cran.r-project.org) using the L-BFGS-B method (Byrd et al. 1995) to calculate the minimum RMSE. During optimization, stochastic effects for sea-level rise, and elevation were removed, and R iteratively solved the model, minimizing RMSE between the model result from 1995 to 2008 and the interpolated 2008 DEM. With a coefficient of 0.04, the RMSE improved from 0.48 to 0.11 m.

We used three sea-level rise projections for the Pacific coast of North America that used GCMs developed for the Intergovernmental Panel on Climate Change Fourth Assessment Report, but incorporated regional factors, including local steric variations, wind-driven differences in ocean heights, and gravitational and vertical land motions (National Research Council 2012). Projections of sea-level rise were allowed to vary by the observed amount of variation from the Golden Gate gauge from 1995 to 2013 (standard deviation [SD] = 0.03), thereby introducing stochastic effects into the elevation model. The elevation model was run 100 times over the three sea-level rise scenarios that represented low, mid, and high (+44, +93, and +166 cm, respectively) by 2100 (National Research Council 2012). All three curves have a positive curvature with acceleration with increased global temperatures in the later part of the century. The average annual sea-level rise curve was used as the input function, and we assumed the difference between the maximum tidal height and minimum tidal height (tide range) remained constant through time, with only MSL changing annually.

Our model is an empirical model that relates tidal inundation of the marsh surface to observed elevation change. We assumed that sediment availability and local processes (e.g., freshwater input, organic matter production, and rates of decomposition and compaction) are constant through time. We included processes that were found relevant to this marsh; erosion at the bay edge was not observed or included (Thorne et al. 2012). Although uncertainty increases when downscaling sea-level rise projections to a small spatial area, the extensive empirical local datasets and model validation with observed data improve confidence in the results.

Results

Site characteristics

Currently, SPBNWR lies above MSL and is dominated by a relatively flat high marsh platform that is only inundated during MHW and higher tide levels. The 1995 survey was composed of 1,369 points, with an elevation range of 2.97 m ($\bar{x}$ = 1.83 m, SD = 0.32; MHW, NAVD88), a minimum elevation of −1.37 m (MHW), and a maximum of 1.59 m (MHW). The elevation survey done in 2008 in the same area contained 649 elevation points, with an overall range of 2.54 m ($\bar{x}$ = 1.84 m, SD = 0.28; MHW, NAVD88), a minimum elevation of −1.05 m (MHW), and a maximum elevation of 1.48 m (MHW). The RMSE of the 1995 and 2008 DEMs was 0.07 and 0.15 m, respectively. Elevation differences between 2008 and 1995 ranged from −1.16 to +1.41 m ($\bar{x}$ = −0.01, SD = 0.15). When DEMs were compared, a spatial pattern was observed where elevation decreases appeared to be located within the interior of the marsh near constructed levees, whereas increases in elevation were associated with the San Pablo Bay edge. Levees appeared to limit sediment transport to the upper reaches of the marsh and reduce any overland flow of water and sediment. Our model comparison showed that the marsh is subsiding in the interior, where 55% of the marsh surface decreased in elevation. The category with a −0.1 to 0 m elevation change represented the largest group of grid cells, encompassing 39% of the cells in the study area (see Thorne et al. 2014).

Vegetation was sampled at 511 locations from the San Pablo Bay edge to the marsh-upland transitional zone along transects. Distinct vegetation patterns were observed in relation to MSL, since plants are typically restricted by their inundation tolerance and soil salinities. The overall plant community had relatively low diversity, with only 10 species recorded and high combined species percent cover (~100%) in most places (Table 2). California cordgrass *Spartina foliosa* was recorded at 59 plots and was located in a narrow band along the San Pablo Bay edge. Pickleweed was present across most of the marsh surface at 85% of the plots (n = 457) where tidal inundation occurs during MHW or MHHW. Salt marsh dodder *Cuscuta salina*, a parasitic plant often found on pickleweed, was found in 25% of the plots. The upper reaches of the marsh farthest from the San Pablo bay edge supported coyote brush *Baccharis pilularis* and alkali heath *Frankenia salina*, typically adjacent to constructed levees. Invasive broadleaved pepperweed *Lepidium latifolium* was recorded at seven plots and was clumped along a tidal rack line near the San Pablo Bay edge. Primary plant species were defined as >70% occurrence within plots. This was used to define the plant community for low, mid, and high-marsh to relate state changes with sea-level rise (Table 3).

Marsh response model

Our empirical marsh sea-level rise response model is based on annual inundation depth and observed elevation change, and resulted in E_t for each grid cell. The model showed a decrease in mean elevation for each grid cell predicted forward at annual steps to the year 2100 (Table 4; Figure 3; Table S1). Mean elevation relative to MSL in 2010 under the moderate sea-level rise scenario was 0.79 m (MSL, SD = 0.20), with a final mean elevation of −0.31 m (MSL, SD = 0.19) in 2100. Under the moderate sea-level rise scenario, the model projected a decrease in elevation to 0.52 m MSL across the entire surface by 2050. Under the mid and high sea-level rise rates, the model projected the entire marsh would be below MSL and therefore drowned between 2080 and 2090 (Figure 3).

We assumed that if E_t <0 MSL, the surface was functionally submerged and therefore no longer able to support marsh plant communities. Plant species elevations relative to MSL (meters) were used to define mudflat; low, mid, and high-marsh; and marsh-upland transition communities (Table 3) and were used to model state changes under the sea-level rise scenarios

Table 2. Vegetation data was collected for species observed within 0.25-m2 quadrat along transects over the marsh surface at San Pablo Bay National Wildlife Refuge in 2008. Vegetation presence related to elevation (m, MSL) was used to define plant communities for modeling and interpretation for impacts to wildlife habitats. Height measurements exclude *Cuscuta salina*, a parasitic plant that lives entangled on *Sarcocornia pacifica*.

Scientific name	Common name	*n*	Elevation (m) relative to MSL[a]					Cover (%)				Average height (cm)				Maximum height (cm)			
			$\bar{x}$	σ	Min.	Max.	Range	$\bar{x}$	σ	Min.	Max.	$\bar{x}$	σ	Min.	Max.	$\bar{x}$	σ	Min.	Max.
Sarcocornia pacifica	Pickleweed	457	0.72	0.16	0.23	1.8	1.57	79	33	1	100	45	12	10	75	54	13	15	86
C. salina	Salt marsh dodder	118	0.83	0.08	0.61	1.04	0.42	27	25	1	95	—	—	—	—	—	—	—	—
Spartina foliosa	California cordgrass	59	0.26	0.32	−0.5	0.64	1.14	37	25	1	85	71	17	30	105	77	16	45	110
Frankenia salina	Alkali heath	47	0.93	0.16	0.54	1.32	0.78	50	33	5	100	22	10	5	50	27	11	5	55
Grindelia stricta	Gumplant	46	0.75	0.13	0.45	1.04	0.59	37	30	1	100	67	16	35	100	73	18	35	115
Distichlis spicata	Salt grass	18	0.89	0.18	0.7	1.53	0.83	54	32	1	100	12	7	5	25	14	9	5	35
Jaumea carnosa	Jaumea	15	0.74	0.13	0.46	0.97	0.51	65	31	25	100	15	10	5	40	18	12	5	50
Baccharis pilularis	Coyote brush	15	1.05	0.29	0.71	1.69	0.98	40	29	5	100	75	29	33	150	85	38	35	200
Lepidium latifolium	Broadleaved pepperweed	7	0.89	0.18	0.54	1.07	0.54	15	11	2	30	48	35	1	85	50	36	1	87
Atriplex triangularis	Fat hen	2	1.09	0.32	0.87	1.32	0.45	26	34	2	50	13	18	1	25	13	18	1	25

[a] MSL = mean sea level.

(Figure 4). Model results indicated plant communities will shift spatially and temporally, changing their habitat availability for marsh wildlife with sea-level rise (Figure 5). For example, marsh-upland transition habitats comprised 39 ha in 2010, but they were projected to be quickly lost to sea-level rise by 2030 under all scenarios. In 2010, high marsh habitats comprised the largest amount of area (253 ha; 78%), but disappeared by 2050–2080, depending on the scenario. The model projected a shift from high to mid marsh habitats throughout the century. Mid marsh comprised 7% (30 ha) in 2010 and is projected to peak at 46% in the 2050s under all scenarios, before converting to low marsh and then mudflat by 2100. In 2010, low marsh only occurred in 4% (18 ha) of the area. However, model results showed a gain through 2075 to 65% (291 ha) before gradually declining through the later part of the century, with none remaining by 2100 under mid and high sea-level rise scenarios (Figures 4 and 5). Expansion of mudflats occurred within the survey area to 2100 under all three scenarios. Originally mudflats comprised 3% (11 ha) of the study area, but they were projected to be the dominant feature under mid and high sea-level rise scenarios, with 100% (324 ha) by 2100. Our model did not consider marsh migration or upslope transition.

Discussion

Marsh response

In this study, we developed a local empirical model that incorporated altered biogeomorphic processes. These processes include accretion and ongoing interior subsidence. Subsidence may be due to indirect effects of levee construction and low channel complexity that may have delivered sediment to the upper reaches of the marsh (Reed et al. 1999; Ganju et al. 2004; Thorne et al. 2014). The difference between mid and high sea-level rise rates determined the timing of habitat state changes; however, under both scenarios the model projected all marsh vegetation loss this century with the area transitioning below MSL. Only under the low sea-level rise scenario does marsh habitat persist to 2100, dominated by low marsh habitat. Under mid and high sea-level rise scenarios, all high marsh and marsh-transition habitat is lost near 2050, with a brief state change to mid and then low marsh habitat through the later part of the century.

We used marsh elevation change over 13 y as a proxy for accretion; this modeling assumes these historic relationships will hold into the future. If subsidence no longer occurred and accretion rates increased, marsh persistence with sea-level rise could be more optimistic. Other factors that could increase marsh accretion and capacity to keep pace with sea-level rise may include increased suspended sediment availability and improved tidal delivery to the marsh surface. Both of which could be augmented by management actions on the ground. In addition, other climate factors, such as climate warming and increased precipitation over the next century, could increase plant productivity, which can also facilitate accretion processes (Kirwan et al. 2009).

Table 3. Representative plant species chosen to illustrate plant communities from recorded elevation related to mean sea level (MSL) during 2008 surveys. These were used to define low, mid, high-marsh, and marsh-upland transition habitats to assess plant community state change with sea-level rise to 2100. This allows the assessment of different habitat types lost or gained with sea-level rise.

			Elevation (m) relative to MSL			
Scientific name	Common name	Community	Mean	SD	Low	High
Spartina foliosa	California cordgrass	Low	0.26	0.32	−0.50	0.64
Jaumea carnosa	Jaumea	Mid	0.74	0.13	0.46	0.97
Sarcocornia pacifica	Pickleweed	Mid-high	0.72	0.16	0.23	1.80
Distichlis spicata	Salt grass	Mid-high	0.89	0.18	0.54	1.32
Frankenia salina	Alkali heath	High	0.93	0.16	0.54	1.32
Baccharis pilularis	Coyote brush	High-upland	1.05	0.29	0.71	1.69

Our modeling results for mid and high sea-level rise scenarios contrast with findings from other sea-level rise modeling efforts for this area. For example, Orr et al. (2003) found that San Pablo Bay high marshes would be sustained with low and moderate sea-level rise to 2100. In addition, Stralberg et al. (2011) projected SPBNWR persistence to 2100 with high available suspended sediment and organic matter input. Projections with the widely used sea-level affecting marshes model (SLAMM) indicated a 10-fold expansion in marsh habitat at SPBNWR by 2100, with a 1.0-m sea-level rise (Clough and Larson 2010). In contrast, our model projects loss of all marsh habitats for the mid and high sea-level rise scenarios, with a transition to mudflats below MSL. These other modeling efforts were based on historical elevation datasets or light detection and ranging (LiDAR), often with limited vertical accuracy. Aerial light detection and ranging is used to capture bare earth; however, a study in San Francisco Bay in a heavily vegetated marsh showed vertical elevation error that ranged from 18 to 23 cm compared with on the ground elevation surveys using a real-time kinematic GPS (Foxgrover et al. 2011). Other modeling done here used assumed accretion and suspended sediment rates for this part of San Pablo Bay (Orr et al. 2003; Stralberg et al. 2011; Swanson et al. 2014). Our site-specific datasets and state change models allowed us to examine site-specific variation at a finer resolution, with much higher local confidence, resulting in drastically different results for this site. In addition, our model accounted for altered biogeomorphic processes that resulted in widespread subsidence and low accretion rates, which may be common in highly modified estuaries. This type of site-specific modeling is useful for resource managers who often make decisions at the scale of wildlife home ranges and habitat zones and are constrained by the amount of time and money to implement management strategies for climate change. Additional components should be incorporated into future modeling efforts to more fully understand the

Table 4. Model results for mean elevation relative to mean sea level (MSL) for the marsh surface at San Pablo Bay National Wildlife Refuge under low (+42 cm), mid (+93 cm), and high (+166 cm) sea-level rise by 2100. Mean elevations <0.2 m were not considered marsh habitat and therefore functionally drowned. The model predicts that under all three sea-level rise scenarios most marsh habitats are lost and became unvegetated mudflat by 2100.

Year	Mean elevation (m) under low SLR	Mean elevation (m) under mid SLR	Mean elevation (m) under high SLR
2010	0.78	0.79	0.79
2020	0.72	0.73	0.75
2030	0.68	0.67	0.68
2040	0.63	0.61	0.59
2050	0.57	0.52	0.48
2060	0.50	0.43	0.32
2070	0.43	0.30	0.10
2080	0.34	0.13	−0.19
2090	0.25	−0.08	−0.56
2100	0.15	−0.31	−0.99

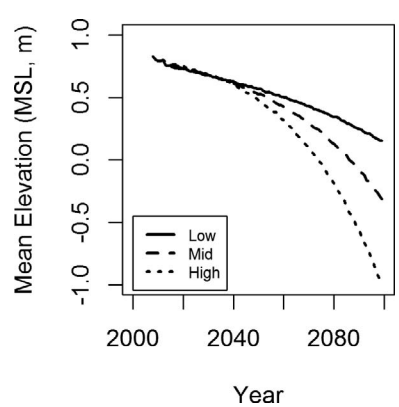

Figure 3. Modeled mean elevation (m, relative to mean sea level [MSL]) for the study site within San Pablo Bay National Wildlife Refuge, located in the northern reach of San Francisco Bay, California, USA, under three sea-level rise scenarios through 2100. Solid line represents low (+42 cm), dashed line represents mid (+93 cm), and dotted line represents high (+166 cm) sea-level rise scenarios by 2100. Elevations below 0.2 m MSL were considered mudflat or subtidal and no longer marsh habitats.

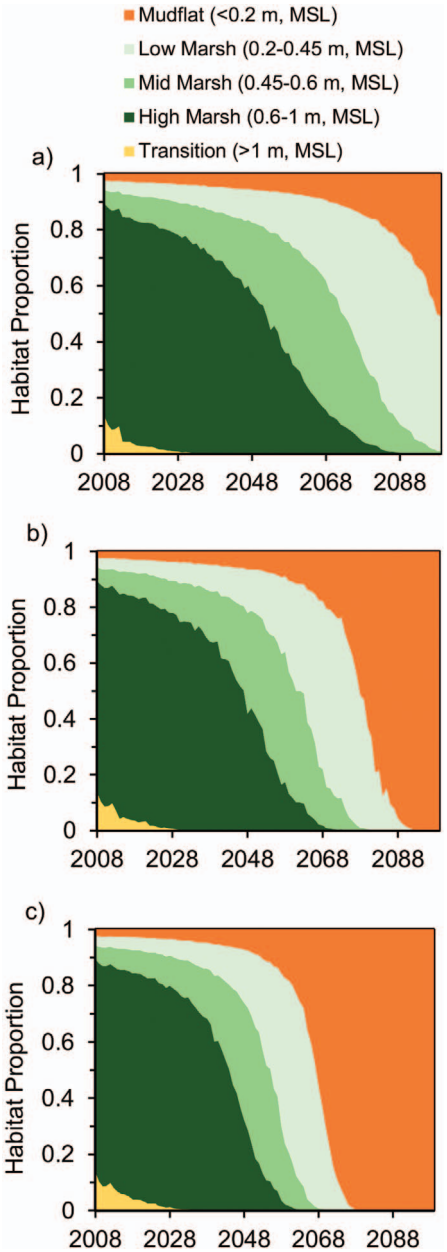

Figure 4. Modeled plant community state changes under (**a**) low +42 cm, (**b**) mid +93 cm, and (**c**) high +166 cm sea-level rise scenarios (National Research Council 2012). Plant communities were defined by elevation ranges (m, mean sea level): marsh-upland transition > 1.0, $0.6 <$ high marsh < 1.0, $0.45 <$ mid marsh < 0.6, $0.2 <$ low marsh < 0.45, and unvegetated mudflat < 0.2. Under mid and high sea-level rise scenarios, the study area transitioned to mudflat between 2070 and 2080, presumably losing all vegetated marsh habitats.

geomorphic and ecological responses, which may reduce the uncertainty in projections. For example, a rise in the groundwater table with sea level could increase flooding events and have negative feedbacks on marsh vegetation and persistence. A better understanding of negative and positive feedbacks between vegetation and accretion rates likely would improve the model results. In addition, although shoreline erosion is currently not occurring, future erosion may have negative impacts on marsh persistence. Lastly, a better understanding of sediment availability and how channel density and complexity play a role in sediment transport into the upper reaches of the marsh would improve modeling efforts. Our study demonstrates a novel methodology to assess marsh responses with sea-level rise by assessing the spatial variability of accretion and inundation across the marsh surface.

Implications for wildlife

Many species' responses to sea-level rise are difficult to predict since their habitat and reproduction requirements are often not well understood. Impacts to terrestrial wildlife may vary over the near term and long term and can include overall loss of habitat availability and protective cover, reproduction success, and access to food (Thorne et al. 2012). Plant communities can be indicative of representative marsh wildlife (Table 1) and can be used as the first step in understanding future species distributions and persistence (Guisan and Thuiller 2005; Veloz et al. 2012). Our models project that SPBNWR marshes would persist under low sea-level rise scenarios; however, the area becomes dominated by low marsh. The modeling predicts that SPBNWR will lose most mid and high-marsh habitats between 2040 and 2060 under mid and high sea-level rise scenarios, presumably due to local subsidence and low accretion rates. Many wildlife species, such as the San Pablo song sparrow *Melospiza melodia samuelis*, a California-listed species of special concern, use high marsh for nesting (Takekawa et al. 2006). Other species, such as the California black rail and salt marsh harvest mouse, have been shown to have limited mobility and can be susceptible to predation during high water events (Evens and Page 1986; Bias and Morrison 1999; Harding et al. 2001), where high marsh and transition-marsh habitats dominated by coyote brush provide refugia (Evens and Page 1986). In addition, many species, such as the northern harrier *Circus cyaneus*, San Pablo vole *Microtus californicus sanpabloensis*, and salt marsh harvest mouse, use mid-high marsh habitat dominated by pickleweed for nesting (Evens et al. 1991; Craig and Beal 1992; Tsao et al. 2009).

We project that low marsh habitats dominated by *Spartina* spp. will persist in portions of the study site until 2080 under all three scenarios. This temporary expansion of low marsh habitats over multiple decades could provide increased habitat for species such as rails if associated marsh features such as channels also develop (Rallidae; Greenberg et al. 2006; Takekawa et al. 2011). In particular, low marsh habitats are critical for the endangered California Ridgway's rails that use *Spartina* spp. for nesting and foraging (Foin et al. 1997). This brief expansion of low marsh habitat over multiple decades could temporarily increase habitat availability for these species, but at the cost of mid and high marsh habitats and their wildlife. Expansion of mudflat habitats could provide foraging areas for many migratory shorebirds (e.g., suborder Charadrii), including the American avocet *Recurvirostra americana* and black-necked stilts

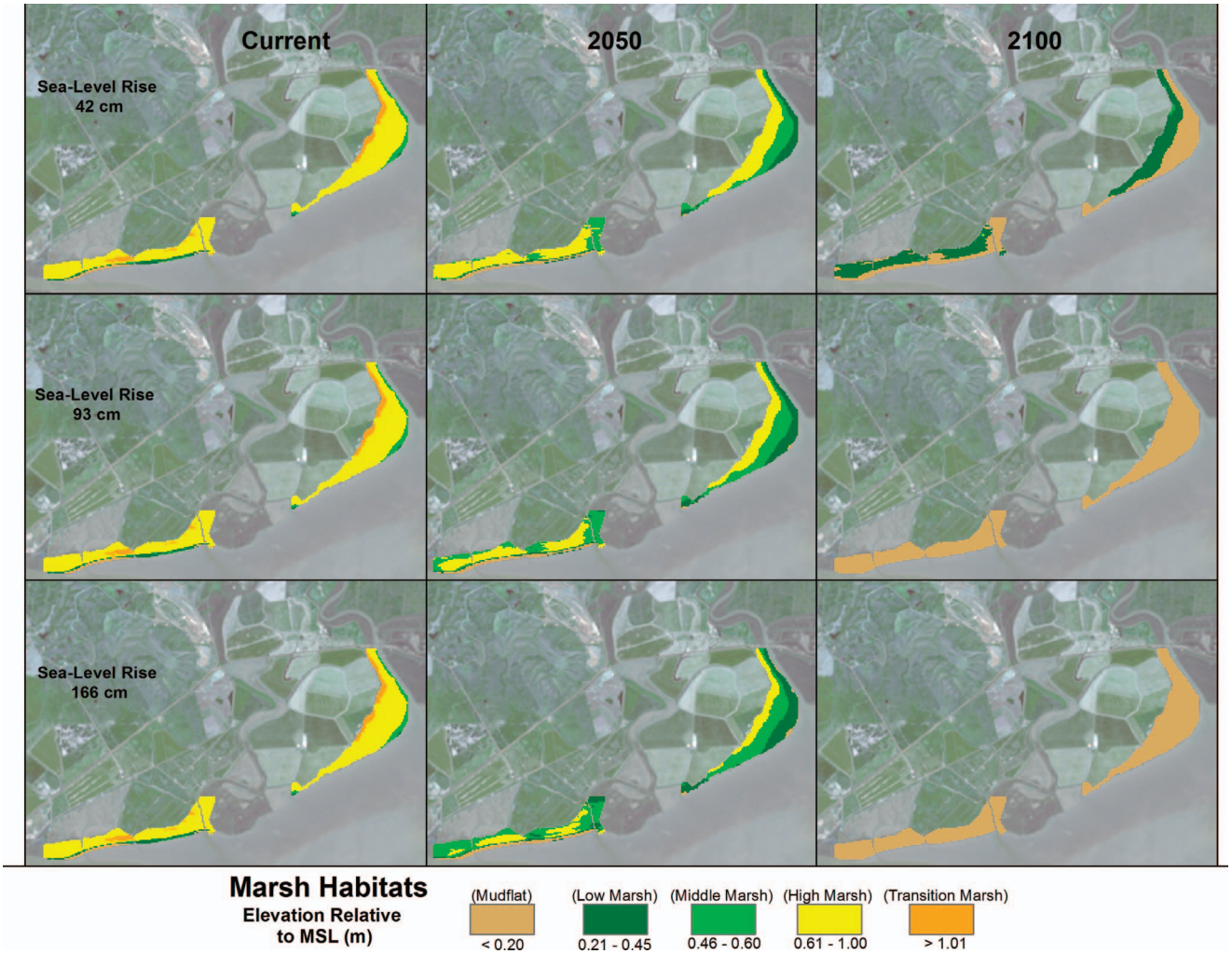

Figure 5. A portion of San Pablo Bay National Wildlife Refuge, located in the northern reach of San Francisco Bay, California, USA, modeled for three sea-level rise scenarios to 2100. Under low sea-level rise rates (+ 42 cm) areas of the study remained in marsh vegetation. Under mid (+ 93 cm) and high (+ 166 cm) sea-level rise rates the study area became unvegetated mudflat by the end of the century. This modeling approach assumes marsh migration upslope would not occur due to levees and infrastructure constraints.

Himantopus mexicanus that rely heavily on mudflats during the winter (Takekawa et al. 2001).

Our results raise concern over the potential for sea-level rise to decrease the spatial extent of these ecologically important marsh plant communities and their associated wildlife in the near term and long term, with little opportunity to relocate to viable habitat in the human-dominated landscape surrounding SPBNWR. However, without rapid management actions, marsh habitat loss will occur if upslope migration cannot occur. The development and implementation of climate change adaption strategies by land managers could prevent the loss of marsh habitats and associated wildlife. Climate change adaptation strategies may include sediment augmentation to increase accretion rates, improved channel complexity, and the development of wildlife refugia from high water and predation pressure. At this study site, marsh migration upslope is not possible due to levee and infrastructure restrictions; therefore, restoration of adjacent lands could increase resilience by increasing habitat area and corridors for dispersal.

Conclusions

The intertidal marsh ecosystems of the San Francisco Bay estuary have decreased but endured over 150 y of modification by humans, resulting in endangered and threatened species receiving state and federal conservation status and the establishment of protected areas (ESA 1973). Climate change impacts will vary by geographical region; however, sea-level rise will impact the coastal zone. Few studies have looked at sea-level rise impacts on marsh wildlife (LaFever et al. 2007; Seavey et al. 2011; Traill et al. 2011), and although these studies are useful, they often lack the amount of site-specific detail needed to make comprehensive

adaptation plans and management strategies. Our empirical sea-level rise response model allowed a finer scale examination of the spatial variability within a marsh that could help identify priority areas for habitat monitoring, restoration, and land acquisition considering future sea-level rise. This novel approach is transferable to other low-lying tidal marsh areas where management decisions are being made at scales meaningful to wildlife. To avoid the potential loss of many marsh-dependent species, management actions need to be adaptive and focused on actions that include future habitat changes. Comprehensive modeling provides valuable insight about local processes and wildlife impacts and improves our ability to implement strategies to reduce biodiversity loss.

Supplemental Material

Please note: *The Journal of Fish and Wildlife Management* is not responsible for the content or functionality of any supplemental material. Queries should be directed to the corresponding author for the article.

Table S1. Data table contains tidal marsh elevations (meters, NAVD88 and relative to MSL) modeled under three sea-level rise projections for low, mid, and high scenarios. Mean elevation results for each scenario for San Pablo National Wildlife Refuge from 2008 to 2099 are presented in the table. Elevation results are in meters (MSL and NAVD88); SD of elevation for each scenario is in meters.

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