Title Page

Title: Wetlands in intermittently closed estuaries can build elevations to keep pace with sea-level rise

Authors: Karen M. Thorne[1*], Kevin J. Buffington[1], Scott F. Jones,[1], and John L. Largier[2]

[1]U.S. Geological Survey, Western Ecological Research Center, One Shields Ave. Davis, CA, 95616
[2]University of California, Davis, Environmental Science & Policy and Coastal & Marine Sciences Institute, P.O. Box 247, Bodega Bay, CA 94923
*Corresponding author, kthorne@usgs.gov, 916-502-2996

Key words: estuary, California, marsh, modeling, climate change

Abstract

Sea-level rise is a threat to coastal ecosystems, which have important conservation and economic value. While marsh response to sea-level rise has been well characterized for perennially open estuaries, bar-built intermittently-closed estuaries and their sea-level rise response are seldom addressed in the literature – despite being common globally. Here, we show that annual closures can play a critical role in maintaining marsh elevations by trapping fluvial sediments that can accrete on the marsh plain. We seek to advance the conceptual understanding of sea-level rise response of marshes by incorporating the unique nature of intermittently-closed estuaries in a marsh model. We hypothesize that intermittently-closed-estuary marshes may be more resilient to sea-level rise than open-estuary marshes due to greater initial elevation capital and higher accretion rates due to closure events. Using California, USA as a case study, spatial analysis shows that marshes in intermittently-closed-estuaries had significantly greater elevations ($\bar{x}$ = 1.93 m ± 0.2 standard error, n = 14) than marshes in permanently open estuaries ($\bar{x}$ = 0.94 m ± 0.1 standard error, n = 8; P = 0.001). We then used a process-based model to determine marsh elevation change under 840 simulated responses to sea-level rise scenarios to 2100. Our modeling shows that regular annual mouth closure can promote accretion rates and increase marsh elevations fast enough to match even high rates of sea-level rise, as fluvial sediment pulses can be captured in the estuary. Modeled suspended sediment concentration had the strongest effect on accretion, followed by probability of annual mouth closure. Intermittently closed estuaries are critical environments where marshes may be sustained under high rates of sea-level rise, thus reducing the anticipated global loss of these important ecosystems. Our results demonstrate an important gap in the knowledge about marsh accretion and identifies research needs to inform coastal management.

1. Introduction

Estuaries are transition spaces between land and sea that are remarkably productive ecosystems and provide important services to human society (e.g., Brander et al. 2006), but are vulnerable to environmental alteration and climate change (Morris et al. 2002; Cahoon et al. 2006; Shile et al. 2014, Thorne et al. 2018). The severity and evidence of sea-level rise is well documented globally; however, sea-level rise rates are largely uncertain in the later parts of this century as they are dependent on global greenhouse gas emission scenarios (Mather et al. 2009, Bonaduce et al. 2016, IPCC 2018, Rojas et al. 2018). Accelerations in sea-level rise and other climate change drivers will create novel and, in many cases, extreme environmental conditions in estuaries with unknown ecosystem consequences.

Estuaries are traditionally thought of as open to the ocean with one or more rivers or streams flowing into them creating a brackish environment. However, in some estuaries the entrance is not permanently open to the ocean but shifts between open and closed states (McSweeney et al. 2017a), a phenomenon that has led to recent redefinition of an ‘estuary’ (Day 1981, Whitfield and Elliott 2012). These bar-built intermittently closed estuaries (ICE, Figure 1) may take many forms but are typically found on wave dominated coasts, characterized by small tidal prisms, high sediment supply and variable river or creek inflow that allow waves to build a sand barrier across the mouth. ICE mouth closure frequency is dependent on local hydrologic, geomorphic, and ocean conditions (Behrens et al 2013, McSweeney et al. 2017a, Kjerfve 1986, 1994). There is a spectrum of closure frequency within and across ICE, with a single seasonal closure being most common and irregular episodic closures observed in some systems (Elwany et al. 1998, Morris and Turner 2010; McSweeney et al. 2017a; Winter 2020). ICE can be disconnected from the ocean for days or even years, creating an impounded lagoon behind the sand barrier with variable water levels depending on the net water budget (Figure 1). Closed ICE water levels can be highly variable which can vary across climatic regions and with seasons or time (Schallenberg et al. 2010, Clark and O’Connor 2019). Water levels in ICE depend on when the closure occurs, local watershed flows, seepage through barrier, wave overwash and evaporation rates (Stretch and Parkinson 2006).

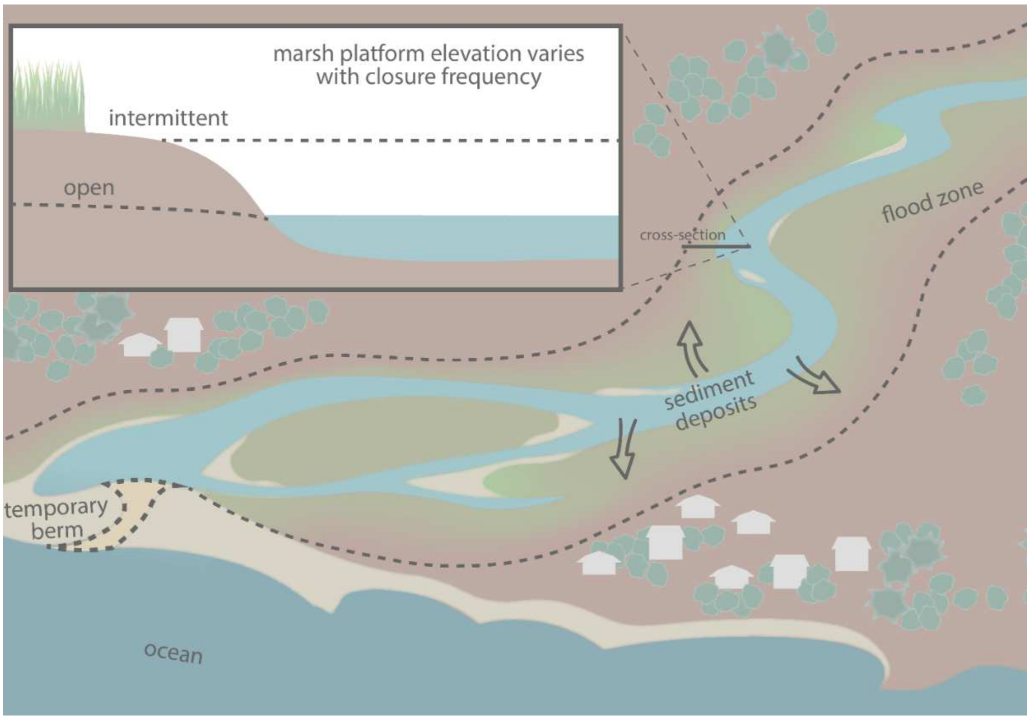

Figure 1. Conceptual model of the biophysical marsh processes and water levels for intermittently closed estuaries (ICE). ICEs are closed to ocean influence a portion of the time usually by a sand-built berm which can be breached by waves and/or high watershed flows. Marshes (green) in ICE have higher marsh platforms when compared with marshes in perennial open estuaries.

The timing and frequency of mouth closure depends on complex interactions between sediment import and export driven by waves, tidal exchange, and river discharge (Whitfield et al. 2012; Behrens et al 2013, 2015). Waves create a berm that can partially or completely block the lagoon inlet. However, if tidal currents or fluvial discharge are large enough (relative to wave forcing), the lagoon inlet will not close. When the mouth is closed, river inflow can fill the basin and lead to overflow and re-opening of the mouth through scouring of a new channel (Rich & Keller 2013; Behrens et al 2015). ICE therefore exhibit closure patterns linked to seasonality in waves and river discharge. During closure events, tidal action and ocean sediment input are halted while river inflows and sediment influx can continue influencing marsh accretion processes. These riverine flows may be the dominant driver of inlet opening or closing on timescales longer than a few years (Elwany et al. 1998).

ICEs are found worldwide and make up over 15% of microtidal (< 2 m tidal range) estuaries (McSweeney et al. 2017b), they are rare in mesotidal regions and absent in macrotidal regions (Figure 2). Often found in Mediterranean-climate zones due to seasonal rainfall patterns and microtidal systems, geographical hotspots include California with over 70% of local estuaries categorized as lagoons (Elwany 2011; Behrens et al 2013, Heady et al. 2014), Australia (Hodgkin and Hesp 1998; Morris & Turner 2010), South Africa (James et al. 2007; de Lecea 2016), Chile (Bertrán et al. 2006), Portugal, and in the Mediterranean Basin itself (Pérez-Ruzafa et al. 2011). California ICEs represent a globally-important resource; in southern CA alone they make up more than 5% of global ICEs, and a larger percentage of Mediterranean-climate ICEs (McSweeney et al. 2017b). California ICE are especially vulnerable ecosystems, as they are small and easy to modify as well as located near rapidly expanding urban centers or agriculture landscapes which can alter watersheds and diversity (Riley et al. 2005; White and Greer 2006). These ICEs are known for their biodiversity and contributions to the economic viability of the surrounding communities (Kwak 1997; Danovaro 2007; Barnes et al. 2008). ICEs are one of the most sensitive estuary types to human activities (Boyd et al. 1992); over the last hundred years, many ICEs have undergone drastic adverse effects from human modifications to the system that include disturbance in water inflow and outflow, mouth stability, runoff from urban areas, salinity balance changes, and invasion of species (Cohen et al. 2005; He and He 2008; Gittman et al. 2015).

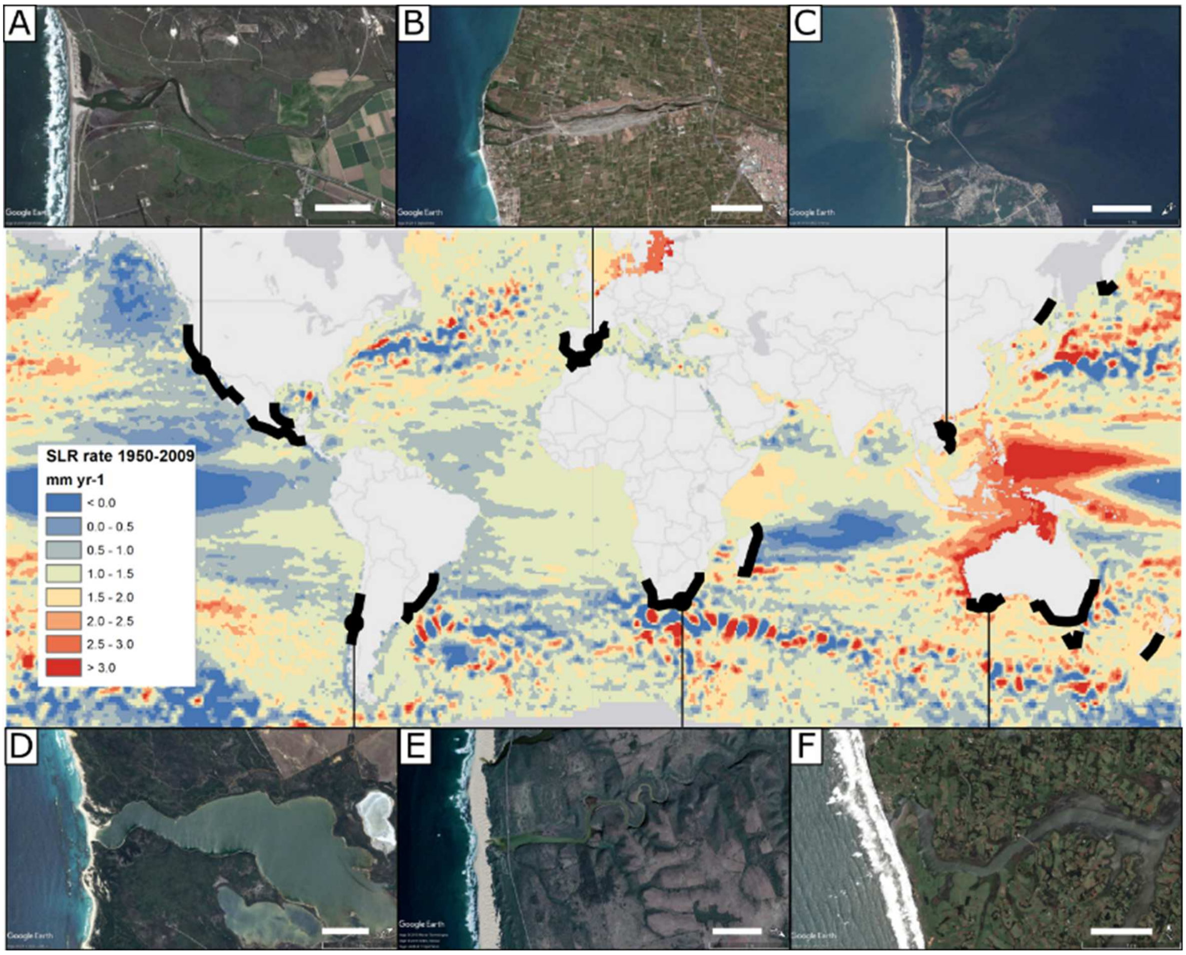

Figure 2. *Intermittently closed estuaries (ICE) occur worldwide and are documented in most microtidal estuaries (black lines adapted from McSweeney et al. 2017b). Sea-level rise rates vary globally making it difficult to interpret impacts to ICEs (dataset modified from Hamlington et al. 2011). A) Santa Ynez, CA (NRC 2012); B) Mijares, Spain (Bonaduce et al. 2016); C) Cau Hai, Vietnam (Tran et al. 2017; D) Lake Budi, Chile (Rojas et al. 2018); E) Mgwalana, South Africa (Mather et al. 2009); F) Stokes, Australia (White et al. 2014). The scale bar represents 1 km.*

The extent of emergent marshes in ICE varies substantially across regions and is largely unknown, but assumed to be related to closure frequency, inundation, salinity, and freshwater inputs (Figure 1). Coastal marsh plants have varying tolerances of inundation and salinity (Schile et al. 2011; Janousek et al. 2016), and their presence or productivity in ICE may be largely influenced by closure frequency and freshwater inputs. Zedler et al. (1980) found that elimination of tidal flow due to lagoon closure decreased primary productivity without freshwater flow; however, if freshwater runoff was present, primary production increased.

Coastal marshes may ‘keep pace’ with sea-level rise through enhanced organic production (Morris et al. 2002) and mineral accretion rates (Kirwan et al. 2010), but high rates of sea-level rise are likely to overwhelm their natural soil building ability leading to submergence over the century (Thorne et al. 2018, Kirwan et al. 2010). Marsh accretion can be composed of mineral contributions from both oceanic and fluvial sources (Reed 1995). There is extensive literature on how marsh biophysical feedbacks in permanently open estuaries can facilitate accretion with sea-level rise (e.g., Reed 1995, Morris et al. 2002, Swanson et al. 2014); however, biophysical feedbacks in ICE marshes are complicated by mouth closure and loss of ocean connectivity, and this type of information is largely missing from the literature. Here, we seek to advance the understanding of marsh processes by incorporating the unique accretion nature of ICE marshes into understanding of sea-level rise vulnerability.

This study aims to use the region of California, USA to evaluate marsh accretion for a range of sea-level rise scenarios in an archetype ICE. We hypothesize that lagoon closure increases marsh accretion regardless of other negative effects of closure (e.g., vegetation biomass loss, compaction). Specifically, we 1) conducted an elevation assessment of marshes in ICEs and permanently open estuaries throughout California, and 2) modeled an ICE marsh to test the sensitivity of accretion to mouth closure frequency to identify the key inputs that control elevation outcomes by 2100 under sea-level rise scenarios. The information gained in this case study can inform the management and scientific understanding of ICE marshes in other settings.

2. **Materials and Methods**

2.1 Case study region

ICE in California have mixed semi-diurnal tides and tend to be shallow with small river/creek inflow. These fluvial inflows are important for opening the mouth following winter rain events, but at other times the mouth state is primarily controlled by tidal prism and closures are driven by wave events (Behrens et al 2013, 2015; Harvey et al 2020). Interannual variability in closure follows rain/river flow cycles as well as mouth management strategies (Elwany et al. 1998; Winter 2020). Salt marsh soils across California estuaries are characterized by mineral sediments and typically have low (<20%) organic matter content (Callaway et al. 2012; Thorne et al 2016; Hinson et al. 2017). Emergent vegetation communities in these marshes are dominated by salt tolerant species such as *Frankenia salina, Jaumea carnosa, Salicornia pacifica* as well as freshwater genus *Alnus sp., Juncus sp.* and *Salix sp.* (Thorne et al 2016, Clark and O'Connor 2019). Summer salinity in California ICEs can be above 25 PSU with water temperatures within portions of the estuary above 30 degree C and dissolved oxygen concentrations can range from 0 mgL^{-1} to 20 mgL^{-1} (Largier et al 2015; Clark and O'Connor 2019).

2.2 ICE marsh characterization

We examined marsh elevation across 22 California estuaries using available Lidar data (2009-2011 NOAA National Ocean Service Office for Coastal Management, CA Coastal Conservancy Coastal Lidar Project) and National Wetland Inventory (NWI) data (Figure 3). We used the NWI database to delineate marshes, assumed that all vegetated estuarine classifications were emergent wetlands, and used their extent to calculate mean elevations from a lidar-derived digital elevation model. We recognize that lidar may not reliably penetrate to the marsh surface

(Buffington et al 2016), however for this analysis we assumed that the vertical bias would be relatively consistent across sites. Additionally, these types of estuaries in California are typically dominated by short-stature vegetation (Thorne et al 2016) so vertical bias can be small. We categorized each estuary as either open year-round to ocean influence or as intermittently closed based on a literature review, expert knowledge, and google earth exploration (Table 1). We compared mean marsh elevation (relative to MSL) between open and intermittent estuary types for California estuaries using two-sample t-tests in R (R Core Team 2016).

Figure 3. *California estuaries included in the marsh elevation meta-analysis.*

__Table 1.__ Site list and local characteristics for California estuaries used in the spatial analysis.

Site	**Estuary Type**	**Watershed Area (km^2)**	**Marsh Area (ha)**	**Mean Marsh Elevation above MSL (m)**	**Marsh elevation (std dev)**
Bolinas Lagoon	Open	49	7	1.43	0.12
Elkhorn Slough	Open	403	363	0.48	0.35
Morro Bay	Open	200	161	0.75	0.36
Mugu	Open	510	499	0.99	0.46
Seal Beach	Open	224	261	0.58	0.31
Upper Newport	Open	394	184	0.73	0.39
Agua Hedionda	Open	77	30	1.21	0.55
Tijuana River	Open	4532	276	1.34	0.69
Russian River	Intermittent	3846	16	1.98	0.82
Pescadero	Intermittent	209	84	1.34	0.19
Scott Creek	Intermittent	77	7	2.12	0.53
Salinas River	Intermittent	8622	92	1.33	0.27
Guadalupa River	Intermittent	4734	32	3.11	0.38
San Antonio Creek	Intermittent	395	2	3.65	1.17
Santa Ynez River	Intermittent	2322	129	2.40	0.63
Goleta Slough	Intermittent	71	70	1.46	0.72
Ventura River	Intermittent	584	2	1.81	0.20
Santa Clara River	Intermittent	4165	63	2.62	0.49
Malibu	Intermittent	196	9	1.34	0.92
Santa Margarita River	Intermittent	404	66	1.54	0.49
San Elijo	Intermittent	496	108	1.24	0.34
Los Penasquitos	Intermittent	197	85	1.10	0.41

2.3 Closure scenario modeling

We modified a 1-D wetland soil cohort model (WARMER, Swanson et al. 2014) to explore tidal marsh elevation responses to lagoon closure with sea-level rise. The adapted WARMER -Lagoon (hereafter WARMER-L) model that we present here considers the dominant above- and belowground processes that control elevation relative to mean sea level and can be summarized with the general equation:

$$Z_{t+1} = Z_t + Q_{zt} + B_{zt} - D_t - R_t - \Delta S_t \qquad (1)$$

where Z_t is wetland elevation relative to mean sea level at time t, Q_{zt} is accumulated mineral deposition at elevation Z between years t and t+1, B_{zt} is accumulated total organic production over the year at elevation Z, D_t is the accumulated decomposition over a year, R_t is accumulated mass-dependent compaction of soil cohorts over a year, ΔS_t is the amount of sea-level rise over the year, and t is time where $\Delta t = (t+1) - t = 1$ year. Vegetation on wetlands slows water velocities through friction, resulting in minimal erosion of the wetland surface (Leonard and Luther 1995; Christiansen et al. 2000; Möller 2006), thus erosion is assumed to be zero in this model. The model captures vertical wetland accretion processes at a given location and we use it to compare responses across a range of closure and sea-level rise scenarios.

We assumed organic matter production was a unimodal function of marsh elevation, with peak biomass occurring at a given elevation, and set to zero at MSL and the elevation of maximum observed tide. We calibrated the amplitude of this function to match empirical accumulation rates derived from soil cores dated with cesium-137 and sampled at a range of elevations. The decomposition and compaction functions were left unchanged from the original description in Swanson et al. (2014). Briefly, decomposition occurs on the labile fraction of organic matter, estimated from soil characteristics, at rates dependent on age (1, 2, or 3+ year) and depth in the soil. Compaction of a soil cohort depends on the overlying mass, with initial and bottom porosity estimated from soil cores provided as model inputs.

The mineral deposition function was adjusted to account for the influence of lagoon closure and fluvial sediment input to the estuary and is the primary factor that accounts for the changes between the scenarios we considered. Deposition followed Marani et al. (2010),

$$Q_{zt} = Q_{st} \qquad (2)$$

where Q_{zt} is the total accumulated mineral deposition (g/m^2) and Q_{st} is the deposition due to settling. Deposition Q_s was calculated at 15-minute temporal resolution and summed over 1 year to give Q_{zt} We omitted direct capture of sediment by vegetation because of a lack of parameters that describe biomass-stem density and diameter relationships for Pacific coast marsh species; in Atlantic coast systems with slow horizontal flows that are typical of marsh flooding, the proportion of total sediment deposition attributed to direct capture is <10% (Mudd et al 2010). During high tides that inundate the marsh and when the marsh is inundated during closure, we assume a continuous sediment supply with a given suspended sediment concentration (C) and the deposition rate (g/m^2/yr) Q_s given as a function of settling velocity w_s,

$$Q_s(t) = w_s C(t) \text{ when marsh is inundated} \qquad (3)$$

$$Q_s(t) = 0 \text{ when marsh not inundated}$$

To account for tidal currents, settling velocity was calculated dynamically from the rate of water level change, such that:

$$w_s = w_x^{dj} \qquad (4)$$

where w_x is the maximum settling velocity (0.0002 m/s), d is the absolute change in water depth (m) over the 15-minute time interval, and j is a decay coefficient (log(0.0002/100)/0.4) estimated by calibration using soil core accumulation rates at Tijuana River estuary and a baseline SSC of 7 mg/L. By using a dynamic settling velocity, we assume that most deposition occurs during slack water and deposition is reduced under greater water velocities that occur during ebb and flood flows. Further, during ebb tides, SSC in the water column is reduced as sediment is deposited on the wetland surface:

$$\frac{dC}{dt} = -\frac{Qs}{h} - \frac{C}{h}\frac{dh}{dt} \qquad (5)$$

where *dh* is the instantaneous change in water level above the marsh surface, and *h* is water level above the marsh surface. Deposition rate $Q_s=0$ when the water level is below the marsh surface. A continuous function of water level elevation was defined from tidal harmonics at the NOAA San Diego tidal gage station (ID: 9410170), representing the typical mixed tidal regime across California. In order to align waters levels to observed, within estuary observations, water elevations were adjusted +15 cm such that mean high water (MHW) was 81.5 cm above MSL and mean higher high water (MHHW) was 92.5 cm above MSL. See Table 2 for more detail on parametrization.

We defined multiple scenarios that potentially affect marsh accretion due to lagoon closure. Closure duration can vary widely across ICEs from days to over a year (Behrens et al. 2015; Winter 2020), depending on the wave climate and freshwater flows. We set closure duration as a constant six months, typical for many California ICEs and altered interannual frequency (i.e., probability of closure in a given year). Berm height was set to 20 cm above initial wetland elevation and increased at a rate equal to sea-level rise for the given scenario; this assumes that marine sand supply is sufficient to build berms across all sea-level rise scenarios. In the “Flood” scenarios, water elevation was held constant at the height of the berm for a given number of days and $C=C_0$, which assumes that wind-wave-driven resuspension of sediment from adjacent mudflats resulted in continuous sediment availability for the duration of the flooding. In the “Pulse” scenarios, we assumed a sediment pulse of varying maximum SSC (C_k = 100, 250, 500, 750, 1000 mg L^{-1}) occurred during the breach event at the end of closure, before the mouth is fully open, as pulses of sediment are typical of the first rainfall event in seasonally arid regions (e.g., Rosencranz et al 2015). The sediment concentration during a pulse event $C_p(t)$ was described by an exponential distribution that mimicked SSC observations at Los Penasquitos during rainfall events, with peak concentration C_k occurring on the day the mouth opened. Specifically,

$$C_p(t) = C_k e^{-mt} \tag{6}$$

where m is a constant of 0.015 and t is time.

To address our primary objective, we explored marsh elevation response to a range of annual lagoon closure probabilities (0, 10, and 100% chance of closure in a given year), linear sea-level rise (2, 4, 6, 8, 10, 12, 15 mm yr^{-1}), baseline sediment supply (2 and 7 mg L^{-1} average annual SSC), “flood” duration (i.e., closed lagoon water level above marsh plain for 3, 6, 12 or 24 days), and sediment pulse scenarios. This was done at an initial marsh elevation (92 cm above MSL) that approximates the average marsh elevation from estuaries that are always open (see Table 1). We compared marsh elevation changes across these simulations: (three closure probabilities (0,10, 100%), seven sea-level rise rates (2, 4, 6, 8, 10, 12, 15 mm yr^{-1}), two baseline SSC (2, 7mg/L), five pulse SSC (100, 250, 500, 750, 1000 mg/L), four flood duration (3, 6, 12, 24 days) = 840 scenarios to assess marsh elevation trajectories over 100 years. Marsh accretion and therefore elevation change occurred during open and closed mouth phases of modeling.

We analyzed the importance of each modeled factor using random forest and the ‘caret’ package in R (R Core Team 2016), including suspended sediment concentration, flood duration, sediment pulse concentration, sea-level rise rate, and closure frequency as model input factors with no interactions between factors.

***Table 2**. Model parameter values for sea-level rise model simulations for WARMER-L. Initial marsh elevation was set to 92 cm above MSL.*

Parameters	**Value**
Organic density[1]	1.14
Mineral density[1]	2.61
Porosity (Top)[2]	0.87
Porosity (Bottom)[2]	0.74
Refractory Organic Matter (%)[2]	7.0
Root:Shoot	2
Peak OM Elevation (cm, MSL)[2]	72.6
Peak OM (g m-2 yr-1)[2]	351
Settling velocity (m s-1)[3]	0.0002
Particle size (µm)[3]	50

[1] Swanson et al. 2014, [2] Thorne et al 2016, [3] Marani et al. 2010

3. **Results**

3.1 Marsh Characterization

Marsh elevation and watershed characteristics were recorded for both open and intermittently closed estuaries throughout California (Table 1). Marsh elevations were nearly 1 m higher in ICE compared with open bar-built estuaries (m above MSL 1.93 ± 0.2 for ICE and 0.94 ± 0.1 for open estuaries; $t = 4.1$, $P = 0.001$; Figure 4). This finding motivates and supports our modeling assumption that accretion occurs during closed mouth periods when water level is above the marsh plain elevation.

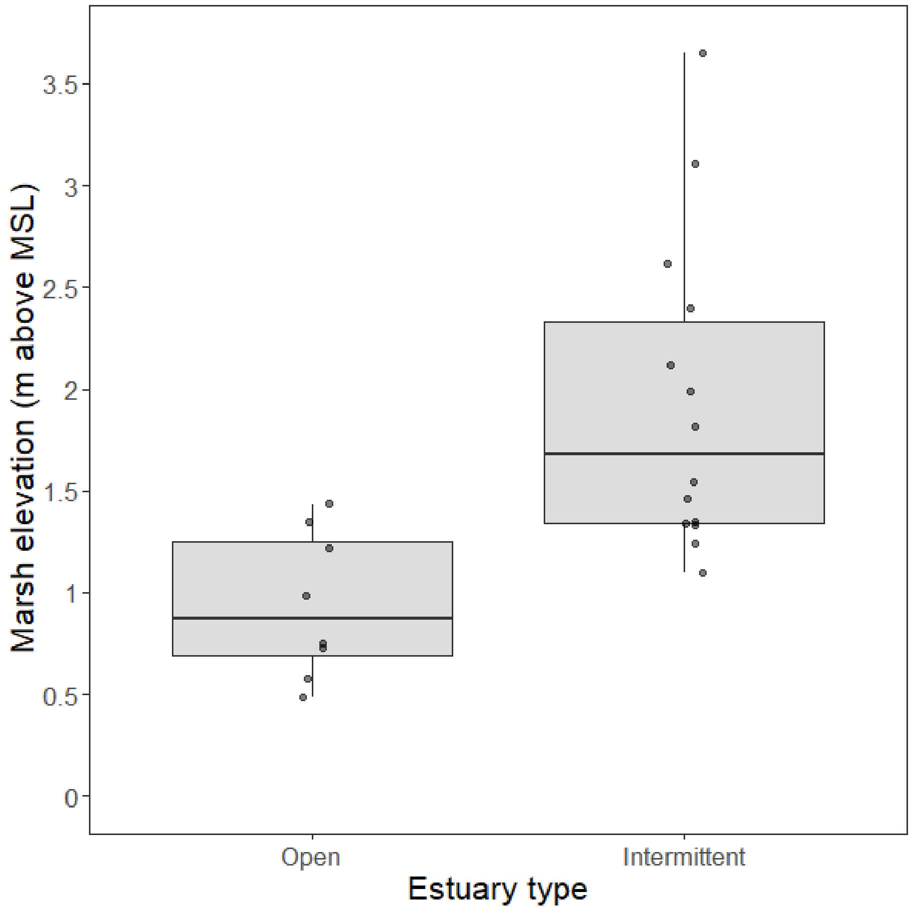

Figure 4. *Average marsh elevation (m above MSL) for California marshes based on mouth type, perennial open estuaries (open) and intermittently closed estuaries (Intermittent, ICE). MSL is at 0 cm. Intermittent marshes are higher in elevation than marshes in always open estuaries (P = 0.003).*

3.2 Closure Scenarios

Using a process-based soil elevation model (WARMER-L) run for 840 scenarios, we determined that lagoon inlet closure, coupled with delivery and trapping of fluvial sediment, can increase marsh elevations relative to sea-level rise rates when compared with estuaries that do not close. More common annual mouth closure and higher sediment delivery resulted in increased marsh accretion rates (Fig. 5) and reduced the overall vulnerability of the marsh platform to submergence from sea-level rise (Fig. 6-8). However, in scenarios with little to no fluvial sediment delivery, closure decreased marsh elevations due to low accretion rates resulting from the exclusion of tidal sediment fluxes (Fig. 8A). Marsh accretion rates in scenarios without closure were not able to build elevations to 'keep pace' with the lowest rates of sea-level rise (Fig. 6), given the assumptions of a low baseline sediment concentration; however, with closure occurring only one year in ten (10% annual closure frequency) the marsh was able to keep pace with slower/historic rates of sea-level rise.

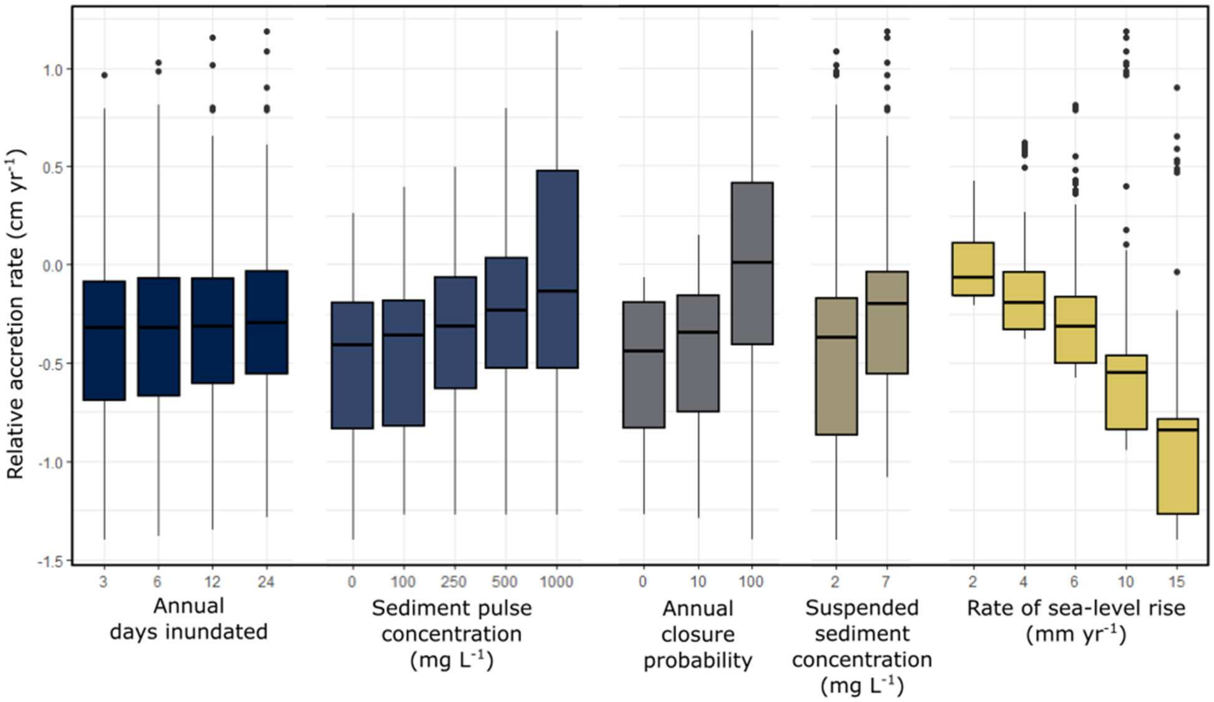

Figure 5. *Average 100-year accretion rate distributions for each model parameter considered individually. Modeled parameters are ordered from least to most important after variable importance calculated using random forest and 'caret' package in R. Accretion rates below zero indicate marshes that are not keeping pace with sea-level rise after 100 simulated years. For each bar with given parameter value, there are multiple simulations due to varying other parameter values. Boxes are the first and third quartiles around the median, whiskers are 1.5x the interquartile range, and dots are outliers.*

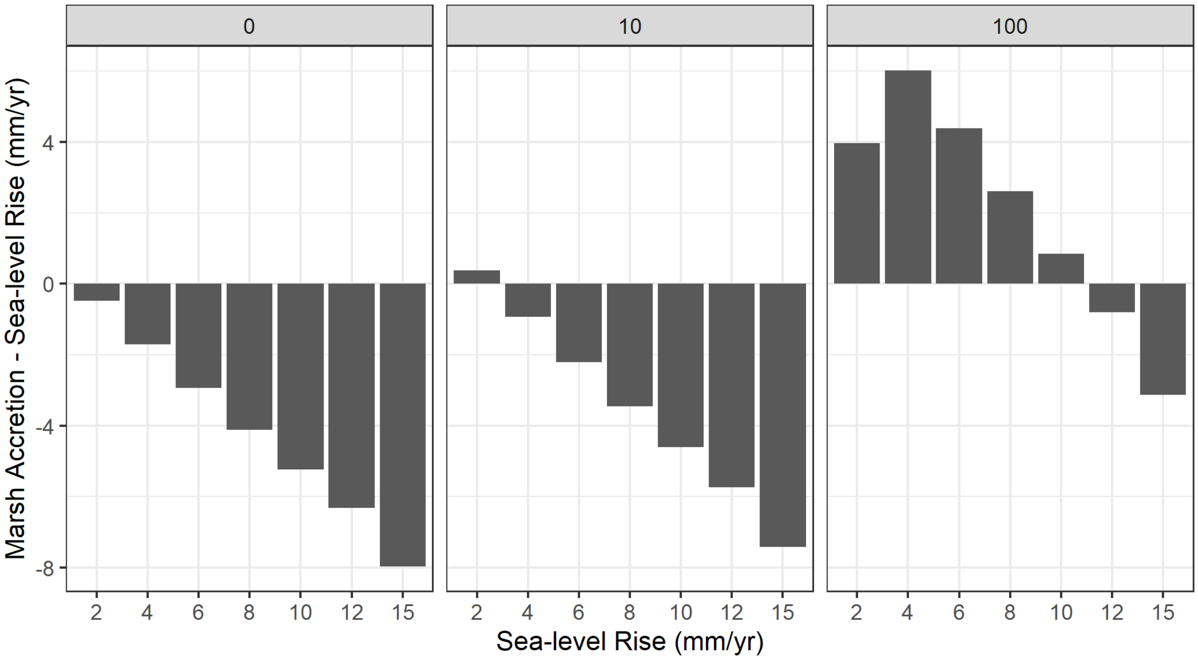

Figure 6. *Marsh accretion rate relative to rate of sea-level rise, across a range of sea-level scenarios, and annual closure frequencies (0, 10, 100%) by 2100. Projected global mean sea level is between 5.4 mm yr^{-1} (RCP2.6) and 15 mm yr^{-1} (RCP8.5) by 2100 (IPCC 2019). Results from the baseline SSC=7 mg/L, flooding days=6, and sediment pulse = 500 mg/L scenario.*

We explored a range of parameter values for, sea-level rise, closure frequency and sediment supply. Using a variable importance analysis, we found that the rate of sea-level rise was the single most important factor in determining changes in the relative elevation of the marsh plain after 100 years. Baseline SSC (55% effect relative to SLR), annual closure frequency (50% effect) and sediment pulse concentration (40% effect) all were moderately important factors in determining marsh elevations by 2100. The number of days flooded (0 effect) was the least important factor for marsh elevation change that we explored.

Closure and fluvial sediment supply interacted to maintain marsh elevation in an ICE under a range of sea-level rise scenarios (Fig. 7). A large sediment pulse associated with high river flow at the time of mouth opening had a positive impact on wetland elevation (Fig. 5) and demonstrates the importance of stochastic sediment delivery events to long-term wetland resilience. In our model, short-lived sediment pulses of ~750 mg L^{-1} each year were enough to offset very high rates of sea-level rise (15 mm yr^{-1}), while smaller sediment pulses facilitated the accretion of marshes under lower sea-level rise scenarios (Fig. 7C and 7F).

Sediment supply effects on accretion rates were largely from sediment pulses that were assumed to occur during storm runoff, which also induces mouth opening processes (Rich & Keller 2013). Baseline suspended sediment concentrations and the duration of marsh inundation during closure had only minor effects on accretion (Fig. 5). In sediment-limited conditions (baseline suspended sediment concentration = 2 mg L^{-1}, no sediment pulse during opening, short wetland inundation = 3 days), sea-level rise overwhelmed marsh accretion processes and closure reduced resilience with total elevations below MSL (Fig. 8A). Sediment-limited conditions could occur when ICEs are opened mechanically and don't have prolonged closure. In sediment-rich conditions (baseline suspended sediment concentration = 7 mg L^{-1}, sediment pulse during opening 500 mg L^{-1}, long marsh inundation = 24 days), closure can lead to increases in marsh elevation enough to 'keep pace' with sea-level rise as high as 10 mm yr^{-1} (Fig. 8C).

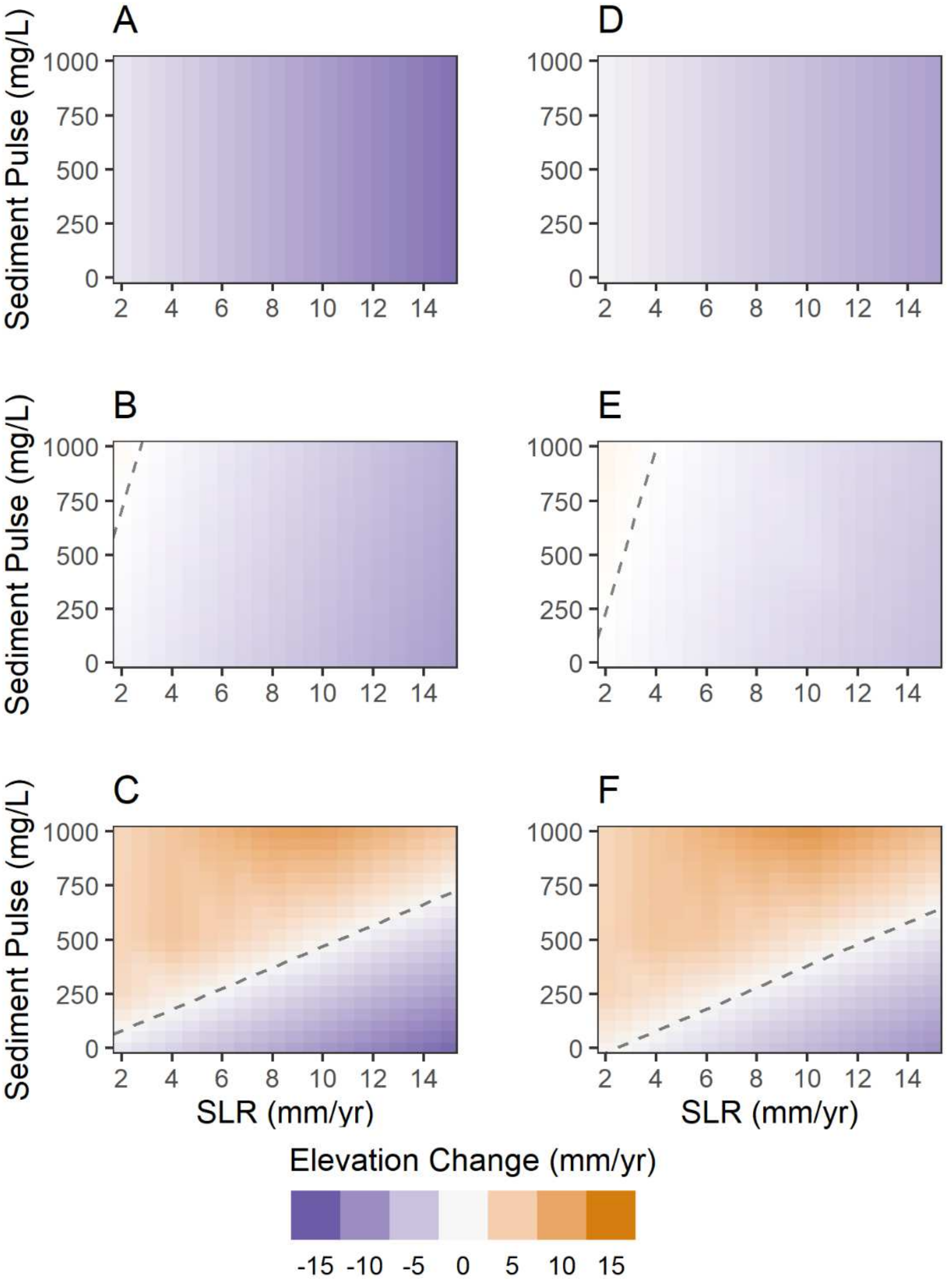

Figure 7. *Average rate of net elevation change after 100-year model simulations across sea-level rise and sediment pulse scenarios for baseline suspended sediment concentrations (SSC) of 2 (A-C) and 7 (D-F) mg* L^{-1}*, 12 days flooded, and an initial elevation of 92 cm above MSL, for an estuary that is always open (A, D), one that has a 10% chance of being closed in a given year (B, E), and one that closes for 6 months every year (C, F). Parameter space above the dashed line shows that positive rates of net elevation change occur in systems with sediment pulses and regular annual closures.*

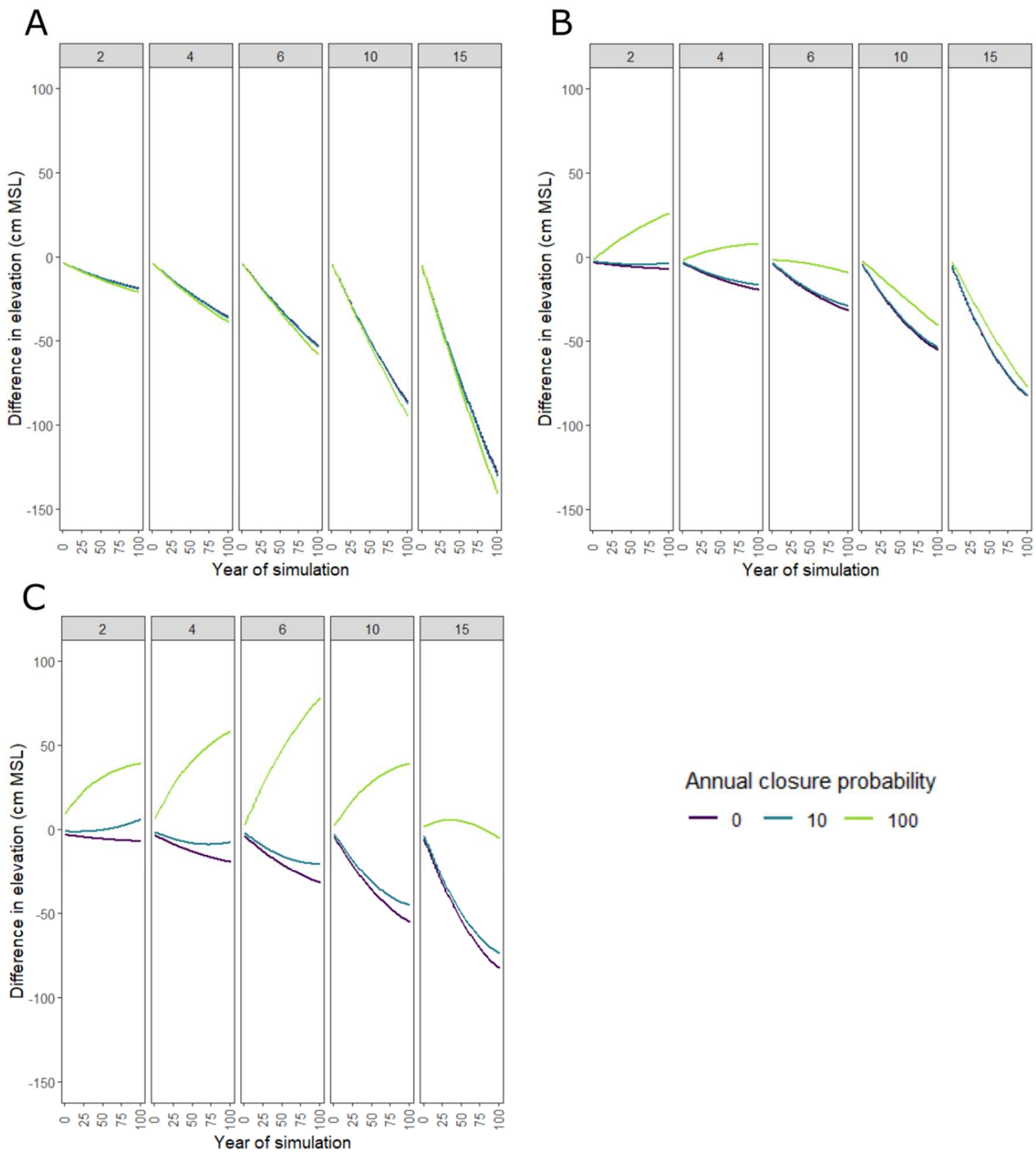

Figure 8. *Difference in simulated elevation from starting elevation over 100 years for estuaries along a closure probability gradient. Overall elevation change over time for A) sediment-starved system (annual days inundated = 3, SSC = 2 mg* L^{-1}*, sediment pulse = 0 mg* L^{-1}*); B) sediment-rich system with no sediment pulses during closure process (annual days inundated = 24, SSC = 7 mg* L^{-1}*, sediment pulse = 0 mg* L^{-1}*); C) sediment-rich system with sediment pulse during closure process (annual days inundated =*

24, SSC = 7 mg L^{-1}, sediment pulse = 500 mg L^{-1}). Gray-titled columns within each panel indicate sea-level rise scenarios. Initial marsh elevation is 92 cm MSL.

4. Discussion

4.1 Marsh evolution

Lagoons with bar-built sand barriers that intermittently close are globally distributed and host important diversity and ecological processes that provide many ecosystem services to human communities. Through spatial analysis we show that marshes in ICE that close frequently have higher elevations than similar marshes in bar-built estuaries that are perennially open. This "elevation capital" has been demonstrated as an important factor in long-term persistence with sea-level rise over the coming century (Cahoon et al. 2019, Cahoon and Guntenspergen 2010). If marshes are unable to maintain their elevations when compared with local tides and sea levels, this can lead to elevation deficits, submergence and loss (Thorne et al. 2018). Accretion of marshes is particularly important in ICE given their tendency to occur in small watershed basins with steep topography and limited upland migration space, this may be especially true for urbanized estuaries (Table 1). The relationships among elevation capital, sea-level rise, and accretion are key components to fully understand long term sustainability of these ecosystems.

Marshes are known to maintain an elevation in equilibrium with sea levels by the accumulation of mineral and organic matter (Morris et al. 2002), which is particularly true in estuaries with continuous tidal exchange. Here we used a process-based model that determined that lagoon inlet closure, coupled with sediment supply, can increase accretion rates in marshes sufficiently to match sea-level rise to 2100 – in contrast to estuaries that are perennially open. Some estuaries would naturally be perennially open, but many estuaries have been armored open to prevent closure processes. Through model simulations we show that ICE marshes can persist under sea-level rise scenarios up to 10 mm/yr when compared with estuaries that didn’t close. However, in scenarios with little to no sediment delivery, closure increased vulnerability of submergence due to the exclusion of tidal flux sediments. In many ICEs more information needs to be collected to better inform these generalizations.

The dominance of mineral sediments in ICE marsh soils highlights the ability of these lagoons to capture sediment either from watershed or oceanic inputs. For example, in California, marshes tend to be mineral-dominated systems, highlighting the importance of sediment input from riverine or ocean sources (Cahoon et al. 1996; Thorne et al. 2016). Also impacts to the watershed from urbanization and development can change sediment availability for accretion process. In our modeling, large sediment pulses that may occur every year when marshes are inundated during ‘natural’ berm-breaching flow events result in accretion that can match high rates of sea-level rise. One study at Seal Beach estuary observed sediment pulses of ~100 mg L^{-1} (400% above the long-term mean) during a relatively mild storm event (Rosencranz et al 2016) demonstrating that stochastic events drive sediment delivery in some areas. It is likely that early winter rainstorms in steep watersheds deliver water and sediment to ICEs at the same time as they are naturally breached, which can supply sediment to the marsh platform. This natural process is weakened if an ICE is manually opened. There is limited observational data regarding the amplitude of sediment pulses to ICEs during these breaching events which are often associated with storms, making it challenging to extrapolate results.

4.2 Climate change

Low watershed flows, micro-tide ranges and high wave energy lead to the presence of ICEs along coastlines. Both terrestrial and ocean conditions are projected to change over the coming decades from climate change (IPCC 2018). In Victoria, southern Australia two ICEs were shown to be mostly governed by relative tide range to determine the mouth state (Kennedy et al. 2020), whereas mouth open or closure state may also be mostly governed by watershed flows (Hinwood and McLean 2015) or by an interaction of watershed flow and wave height (Behrens et al 2013). Climate change projections for California include more flashy precipitation with extreme dry and wet conditions (Dettinger 2013; Polade et al. 2017; Guirguis et al. 2018, Swain et al. 2018), making it difficult to predict future changes in mouth closure state and watershed inflow amounts to these estuaries.

Changes or intensification of storms such as Atmospheric Rivers and El Nino-Southern Oscillation (ENSO), which are important drivers of ICE closures and breaches, could also change the frequency or duration of closures and inflow/sediment influx amounts (Clarke et al. 2017; McSweeney et al. 2017b). Stronger storms with increased wave energy could lead to more frequent ICE closures; in southern California an El Nino winter had elevated ocean levels, larger waves, and low precipitation amounts resulting in a greater number of ICE closures (Harvey et al. 2020). Our results provide insight into how marsh elevations may respond to these closure frequency changes, however, future research linking both sea-level rise and changes in storm precipitation to ICE closure and marsh accretion is important.

Shifts in ICE closure frequency and duration, changes in tidal prism. and salinity regimes may have significant impacts on marshes biogeomorphic processes due to feedbacks among vegetation and soil building processes. Vegetation is a critical component of soil stability and accretion processes that allow a marsh to build soils vertically to prevent submergence with sea-level rise (Morris et al. 2002; Gedan et al. 2011; Kirwan and Guntenspergen 2012). A meta-analysis of ICE from Australia, South Africa, and New Zealand showed that currently ICE had significantly lower salinity and mean annual watershed discharge when compared with open estuaries (Lill et al. 2013), highlighting the importance of freshwater inflow and tidal exchange. Changes in freshwater flow and tidal inundation can alter ICE plant communities, especially in areas with flashy weather and frequent mouth closures (e.g., Zedler et al. 1986, Pezeshki 2001). If increasing closure is accompanied by decreasing freshwater input, estuary water levels may drop due to net evaporation and salinities increase so that plant communities may be altered or lost owing to salinity stress, as in high-salinity marshes surrounding the Mediterranean Sea (Ibnez et al. 2002). Conversely, if increasing ICE closure is accompanied by increasing freshwater inflow, plant communities may shift to those more tolerant to frequent submergence and associated flooding stress (DeLaune et al. 1987), with changes in underlying stress gradients (Morzaria-Luna and Zedler 2014). Loss of vegetation biomass often causes erosion, through scour processes and weaker soils (Gyssels et al. 2005), creating positive feedbacks with increased inundation. However, tidal eroded sediment can be redistributed, enhancing accretion in other locations. Our analysis of the elevations of marshes in ICE and perennial open estuaries illustrates the high elevation perched nature of ICE marshes. Perched marshes can become hypersaline through evaporation after saline inundation, resulting in decreased organic production and loss of plant cover. Conversely, a pulse of freshwater can flush salts from the soils, promoting organic production. Projected changes of warming air and less precipitation across California (Bedsworth et al. 2018) could create negative organic feedbacks in these high elevation perched marshes.

With increasing closure frequency, and therefore environmental stress from salinity and inundation, biological communities in ICE can show changes in diversity of benthic and fish species (Young et al. 1997, Hodgkin and Hesp 1998), altered plankton density and assemblages

(Ortega-Cisneros et al. 2014), and/or increased production for fish (Pollard 1994) and nuisance algal blooms (McLaughlin et al. 2014). As closure frequency increases, marsh area and health may also decrease (Hodgkin and Hesp 1998). There is a lack of studies on how changing environmental conditions within ICE could cause cascading impacts to biotic communities.

Uncertainties in the acceleration and magnitude of global mean sea-level is related to potential ice mass loss and emission scenarios which can make decision making by managers difficult (Haasnoot et al., 2020, Oppenheimer et al. 2019, IPCC 2019). Here, we addressed a range of sea-level rise scenarios (2 – 15 mm yr^{-1}) which encompasses the current projected rates for Representative Concentration Pathways (RCP) emission scenarios (Oppenheimer et al. 2019) and the average current rate of sea-level rise of 2 mm yr^{-1} for California (Board and National Research Council, 2012). IPCC (2019) projects global mean sea level will have an average increase of 5.4 mm yr^{-1} (RCP2.6) to 15 mm yr^{-1} (RCP8.5) to 2100. Our results demonstrated that ICE marshes without mouth closures are at risk of loss under all sea-level rise scenarios (2 – 15 mm yr^{-1}) by 2100; but with annual mouth closure the marshes had accretion rates higher then sea-level rise rates up to 10 mm yr^{-1} (Figure 6). However, sea-level rise will continue to rise and accelerate over the coming centuries posing challenges to ICE worldwide. Thus, the method and results presented here should be reassessed as new research emerges regarding sea levels and atmospheric warming.

4.3 Human impacts and management intervention

Global coastlines and ICEs have been impacted by human development and modifications to the landscape (Kent and Mast 2005), but they continue to provide important biological and economic resources to local communities. Impacts to ICE coastlines include land development, nutrient and pollution runoff, and hardening of shorelines (Page et al. 1995). Penasquitos Lagoon, California is an ICE that closes annually; historically freshwater would have only flowed into the estuary during winter months, but due to urbanization the lagoon experiences year round freshwater runoff and flows (Williams et al. 1998). This freshwater and nutrient inflow has transitioned the plant community from salt tolerant emergent species to freshwater wetlands and riparian ecosystems (Williams et al. 1998). Watershed flow plays a critical role on the seasonal and episodic ICE opening and closures processes, with some ICE experiencing reduced freshwater inflow due to water capture upstream by dams or weirs, water diversion and extraction, or sediment infilling of the estuary (Zedler 1996). Hydrological changes to ICE can alter the marsh accretion processes and mouth closure predictability.

Many estuaries have been hardened to stabilize the mouth opening to prevent closing of the lagoon to reduce localized flooding and other non-desirable conditions, such as eutrophic conditions (Heady et al. 2015). However, our results highlight the threat of sea-level rise for marshes that do not have closure potential – as well as those that are hardened by human development that prevent natural closure processes (Kent and Mast 2005, Thorne et al. 2018). ICEs are often opened mechanically before lagoon water levels rise to the level that would occur prior to natural breaches (Kraus et al. 2008, Largier et al 2019), therefore reducing the number of days it is closed which will reduce marsh flooding and the trapping of watershed sediments for accretion. For example, the Russian River ICE in northern California is mechanically opened as a flood control measure which reduces the inundation of the marsh (Behrens et al. 2013). In contrast, persistent ICE closure in hot/dry conditions of southern California can create low water levels that do not flood the marsh plain (Clark and O’Connor 2019). Additional, management concerns of prolonged closure include eutrophic conditions and poor water quality (e.g., low dissolved oxygen, nutrient laden [Hadwen et al. 2007, Largier et al 2015, 2018; Crooks et al 2018] nuisance algal blooms [McLaughlin et al. 2014; Fong and Zedler 2000]), lowland flooding (Orescanin and Schooler 2018), loss of benthic and fish species (Wooldridge 1994; Hodgkin and Hesp 1998), and increased mosquito populations that are disease vectors (Gersberg et al. 1995). However, mechanical opening of lagoon mouths during a closures phase may reduce available sediment for marsh accretion and reduce the resiliency to sea-level rise.

Site-specific studies are needed to fully understand the tradeoffs between marsh-accretion benefits of allowing ICEs to remain closed and the mitigating negative impacts of human alterations that tidal flushing provides (Largier et al 2019). There are several key processes that require further study and would improve understanding and projections. A better understanding is needed of how sea-level rise and changes in storm intensity will affect: (i) mouth closure dynamics; this is a primary determinant on marsh accretion; (ii) the probability, intensity and timing of sediment pulses that fill the estuary prior to mouth breaching; (iii) how the gradient of mouth closure types (i.e., fully tidal, muted tidal, perched, or closed) may influence flooding and accretion across marshes; and (iv) how vegetation type and density will change with shifts in ICE closure frequency and duration. Also, the tendency to mechanically open ICE to reduce biological and societal impacts should be weighted with possible negative impacts to marsh accretion processes.

5. Conclusion

Our modeling approach provides insight into how the elevation of ICE marshes may respond to sea-level rise under a range of scenarios. We demonstrate that inlet closures can increase marsh accretion and build resilience to sea-level rise and prevent submergence, when coupled with sediment delivery events. When considering sea-level rise, maintaining the possibility of inlet closure and ‘natural’ breach events is important to allow water levels to rise and inundate high marshes, which is necessary for building or maintaining marsh elevations and thus resilience. This implies a need to assess the trade-offs in how land-use management affects tidal prism and marsh processes. Our results indicate an important topic for management consideration and further research needs.

Acknowledgments

Funding for this project was provided by the NOAA National Centers for Coastal Ocean Science Ecological Effects of Sea Level Rise program (NA16NOS4780205). This research was also supported by California State Coastal Conservancy, U.S. Geological Survey Western Ecological Research Center, and University of California, Davis Center for Aquatic Biology and Aquaculture, and the John Muir Institute of the Environment. Valuable input was provided by multiple peer-reviewers, including S. Winter, D. Behrens, D. Elam and anonymous reviewers. Any use of trade, product, or firm names in this publication is for descriptive purposes only and does not imply endorsement by the U.S. government.

References

Bedsworth L, Cayan D, Franco G, Fisher L, Ziaja S. 2018. *California's Fourth Climate Change Assessment: Statewide Summary Report*. California Governor's Office of Planning and Research, Scripps Institution of Oceanography, California Energy Commission, California Public Utilities Commission. Publication number: SUM¬CCCA4-2018-013.

Behrens DK, Bombardelli FA, Largier JL,Twohy E. 2013. Episodic closure of the tidal inlet at the mouth of the Russian River — A small bar-built estuary in California. *Geomorphology* **18**: 66–80.

Behrens D, Brennan M, Battalio B. 2015. A quantified conceptual model of inlet morphology and associated lagoon hydrology. *Shore & Beach* **83**(3): 33-42.

Bertrán C, Vargas-Chacoff L, Peña-Cortés F, Mulsow S, Tapia J, Hauenstein E, Schlatter R, Bravo A. 2006. Benthic macrofauna of three saline-lake wetlands on the coastal rim of southern Chile. *Ciencias Marinas* **32**: 589–96.

Board, O.S. and National Research Council, 2012. *Sea-level rise for the coasts of California, Oregon, and Washington: past, present, and future*. National Academies Press.

Bonaduce, A, Pinardi, N, Oddo, P, Spada, G, Larnicol, G. 2016. Sea-level variability in the Mediterranean Sea from altimetry and tide gauges. *Climate Dynamics*, **47**(9-10): 2851-2866.

Brander LM, Florax RJ, Vermaat JE. 2006. The empirics of wetland valuation: a comprehensive summary and a meta-analysis of the literature. *Environmental and Resource Economics* **33**: 223–50.

Buffington KJ, Dugger B, Thorne K, Takekawa J. 2016. Statistical correction of lidar-derived digital elevation models with multispectral airborne imagery in tidal marshes. *Remote Sensing of Environment*, 186.

Cahoon DR, Guntenspergen GR. 2010. Climate change, sea-level rise, and coastal wetlands. *National Wetlands Newsletter* **32**(1): 8-12.

Cahoon DR, Hensel PF, Spencer T, Reed DJ, McKee KL, Saintilan N. 2006. Coastal wetland vulnerability to relative sea-level rise: wetland elevation trends and process controls. In: *Wetlands and Natural Resource Management*: 271-292. Springer, Berlin, Heidelberg.

Cahoon DR, Lynch JC, Roman CT et al. 2019. Evaluating the Relationship Among Wetland Vertical Development, Elevation Capital, Sea-Level Rise, and Tidal Marsh Sustainability. *Estuaries and Coasts* **42**, 1–15

Callaway JC, Borgnis E, Turner R, Milan C. 2012. Carbon sequestration and sediment accretion in San Francisco Bay tidal wetlands. *Estuaries and Coasts* **35**: 1163-1181.

Christiansen T, Wiberg P, Milligan T. 2000. Flow and sediment transport on a tidal salt marsh surface. *Estuarine, Coastal and Shelf Science* **50**: 315–31.

Clark R, O'Connor K. 2019. A systematic survey of bar-build estuaries along the California coast. *Estuarine, Coastal and Shelf Science* **226**: 106285.

Clarke DW, Boyle JF, and Plater AJ. 2017. Particle-size evidence of barrier estuary regime as a new proxy for ENSO climate variability. *Earth Surface Processes and Landforms* **42**: 1520–34.

Cohen A, Harris L, Bingham B, Carlton JT, Chapman JW, Lambert CC, Lambert G, Ljubenkov JC, Murray SN, Rao LC, Reardon K. 2005. Rapid assessment survey for exotic organisms in southern California bays and harbors, and abundance in port and non-port areas. *Biological Invasions* **7**: 995–1002.

Crooks J, McCollough J, Almeida M, Uyeda K. 2018. *The Physical, Chemical and Biological Monitoring of Los Penasquitos Lagoon* July 2017 - June 2018. Report to Los Penasquitos Lagoon Foundation, 63pp.

Day J.H. 1981. The nature, origin and classification of estuaries. In: *Estuarine Ecology: With Particular Reference to Southern Africa*: 1-6. Balkema, Rotterdam.

DeLaune R, Pezeshki S, and Patrick Jr W. 1987. Response of coastal plants to increase in submergence and salinity. *Journal of Coastal Research*: 535–46.

de Lecea A, Green AN, Cooper A. 2016. Environmental change during the Pleistocene and Holocene: Estuaries and lagoons of southern Africa. In: *Quaternary environmental change in Southern Africa: Physical and human dimensions*: 219-233. Cambridge University Press.

Dettinger MD. 2013. Atmospheric rivers as drought busters on the US West Coast. *Journal of Hydrometeorology* **14**: 1721–32.

Elwany M. 2011. Characteristics, restoration, and enhancement of southern California lagoons. *Journal of Coastal Research* **Special Issue 59**: 246–55.

Elwany MHS, Flick RE, Aijaz S. 1998. Opening and closure of a marginal southern California lagoon inlet. *Estuaries* **21**: 246–54.

Fong P, Zedler J. 2000. Sources, sinks, and fluxes of nutrients (N + P) in a small highly modified urban estuary in southern California. *Urban Ecosystems* **4**: 125-144.

Gedan KB, Kirwan ML, Wolanski E, *et al.* 2011. The present and future role of coastal wetland vegetation in protecting shorelines: answering recent challenges to the paradigm. *Climatic Change* **106**: 7–29.

Gersberg RM, Matkovits M, Dodge D, McPherson T, Boland JM. 1995. Experimental opening of a coastal California lagoon: effect on bacteriological quality of recreational ocean waters. *Journal of Environmental Health* **58**(2): 24-30.

Gittman RK, Fodrie FJ, Popowich AM, Keller DA, Bruno JF, Currin CA, Peterson CH, Piehler MF. 2015. Engineering away our natural defenses: an analysis of shoreline hardening in the US. *Frontiers in Ecology and the Environment* **13**: 301–307.

Guirguis K, Gershunov A, Cayan DR, and Pierce DW. 2018. Heat wave probability in the changing climate of the Southwest US. *Climate Dynamics* **50**: 3853–64.

Gyssels G, Poesen J, Bochet E, and Li Y. 2005. Impact of plant roots on the resistance of soils to erosion by water: a review. *Progress in Physical Geography* **29**: 189–217.

Hadwen WL, Arthington AH. 2007. Food webs of two intermittently open estuaries receiving ^{15}N-enriched sewage effluent. *Estuarine, Coastal and Shelf Science* **71**(1-2): 347-358.

Harvey M, Giddings SN, Stein ED, Crooks JA, et al 2020. Effects of elevated sea level and storm waves on southern California estuaries during the 2015-2016 El Nino, *Estuaries and Coasts* **43**: 256-271.

He L-ML, He Z-L. 2008. Water quality prediction of marine recreational beaches receiving watershed baseflow and stormwater runoff in southern California, USA. *Water research* **42**: 2563–2573.

Heady WN, O'Connor K, Kassakian J, Doiron K, Endris C, Hudgens D, Clark RP, Carter J, Gleason MG. 2014. *An Inventory and Classification of U.S. West Coast Estuaries*. The Nature Conservancy, Arlington, VA. 81pp.

Heady WN, Clark RP, O'Connor K, Clark C, Endris C, Ryan S, S Stoner-Duncan. 2015. Assessing California's bar-built estuaries using the California Rapid Assessment Method. *Ecological Indicators* **58**:300-310.

Hinson AL, Feagin R, Eriksson M, Najjar R, Herrmann M, Bianchi T, Kemp M, Hutchings J, Crooks S, Boutton T. 2017. The spatial distribution of soil organic carbon in tidal wetlands soils of the continental United States. *Global Change Biology* **23**: 5468-5480.

Hinwood JB, McLean EJ. 2015. Predicting the dynamics of intermittently closed/open estuaries using attractors. *Coastal Engineering* **99**: 64-72.

Hodgkin EP, Hesp P. 1998. Estuaries to salt lakes: Holocene transformation of the estuarine ecosystems of south-western Australia. *Marine and Freshwater Research* **49**: 183–201.

Ibnez C, Curco A, Jr. JW Day, and Prat N. 2002. Structure and productivity of microtidal Mediterranean coastal marshes. In:Weinstein, M.P. and Kreeger, D.A. (Ed). Concepts and Controversies in Tidal Marsh Ecology.

IPCC, 2018: *Global warming of 1.5°C. An IPCC Special Report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty* [V. Masson-Delmotte, P. Zhai, H. O. Pörtner, D. Roberts, J. Skea, P.R. Shukla, A. Pirani, W. Moufouma-Okia, C. Péan, R. Pidcock, S. Connors, J. B. R. Matthews, Y. Chen, X. Zhou, M. I. Gomis, E. Lonnoy, T. Maycock, M. Tignor, T. Waterfield (eds.)].

IPCC, 2019: IPCC Special Report on the Ocean and Cryosphere in a Changing Climate [H.-O. Pörtner, D.C. Roberts, V. Masson-Delmotte, P. Zhai, M. Tignor, E. Poloczanska, K. Mintenbeck, A. Alegría, M. Nicolai, A. Okem, J. Petzold, B. Rama, N.M. Weyer (eds.)]. In press.

James NC, Cowley PD, Whitfield AK, Lamberth SJ. 2007. Fish communities in temporarily open/closed estuaries from the warm-and cool-temperate regions of South Africa: A review. *Reviews in Fish Biology and Fisheries* **17**: 565–80.

Kennedy DM, McSweeney SL, Mariani M, Zavadil E. 2020. The geomorphology and evolution of intermittently open and closed estuaries in large embayments in Victoria, Australia. *Geomorphology* **350**: 106892.

Kent B, Mast J. 2005. Wetland change analysis of San Dieguito Lagoon, California, USA: 1928-1994. *Wetlands* **25**:780.

Kirwan ML, Guntenspergen GR. 2012. Feedbacks between inundation, root production, and shoot growth in a rapidly submerging brackish marsh. *Journal of Ecology* **100**: 764–70.

Kirwan ML, Guntenspergen GR, D'Alpaos A, Morris JT, Mudd SM, Temmerman S. 2010. Limits on the adaptability of coastal marshes to rising sea level. *Geophysical Research Letters* **37**: L23401.

Kraus NC, Patsch K, Munger S. 2008. Barrier beach breaching from the lagoon side, with reference to Northern California. *Shore & Beach* **76**(2): 33-43.

Largier, J., D. Behrens, K. Hewett, M. Koohafkan, M. Robart, D. Dann, and R. Roettger, 2020. Managing the mouth of the Russian River Estuary, California. In: *Complex Coastal Systems – Transdisciplinary Learning on International Case Studies*. J.Slinger, S. Taljaard, F. d'Hont (eds), Delft Academic Press, pp 71-86.

Largier J, O'Connor K, Clark R. 2019. *Considerations for Management of the Mouth State of California's Bar-built Estuaries*. Report to Pacific States Marine Fisheries Commission and NOAA/NMFS. 57pp.

Largier, J. L., R. E. Roettger and K. M. Hewett. 2018. Dissolved Oxygen in Pescadero Lagoon. Prepared for Regional Water Quality Control Board.

Largier, J. L., I. W. Aiello, D. Jacobs, J. Lacy, C. Pallud, M. T. Stacey, S. M. Carlson, E. Huber and C.M. Bowles. 2015. Report of Pescadero Lagoon Science Panel. Prepared for California State Parks, 48pages.

Leonard LA, Luther ME. 1995. Flow hydrodynamics in tidal marsh canopies. *Limnology and oceanography* **40**: 1474–84.

Lill AWT, Schallenberg M, Lal A, Savage C, Closs GP. 2013. Isolation and connectivity: Relationships between periodic connection to the ocean and environmental variables in intermittently closed estuaries. *Estuarine, Coastal and Shelf Science* **128**: 76-83.

Marani M, D'Alpaos A, Lanzoni S, Carniello L, Rinaldo A. 2010. The importance of being coupled: Stable states and catastrophic shifts in tidal biomorphodynamics. *Journal of Geophysical Research: Earth Surface* **115**: F4.

Mather A, Garland G, Stretch D. 2009. Southern African sea levels: corrections, influences and trends. *African Journal of Marine Science*, **31**(2): 145-156.

McLaughlin K, Sutula M, Buss L, Anderson S, Crooks J, Dagit R, Givson D, Johnston K, Stratton L. 2014. A regional survey of the extent and magnitude of eutrophication in

Mediterranean estuaries of southern California, USA. *Estuaries and Coasts* **37**:259-278.

McSweeney S, Kennedy D, Rutherfurd I. 2017a. A geomorphic classification of intermittently open/closed estuaries (IOCE) derived from estuaries in Victoria, Australia. *Progress in Physical Geography: Earth and Environment* **41**(4): 421-449.

McSweeney S, Kennedy D, Rutherfurd I, Stout J. 2017b. Intermittently Closed/Open Lakes and Lagoons: Their global distribution and boundary conditions. *Geomorphology* **292**: 142–152.

Morris JT, Sundareshwar P, Nietch CT, Kjerfve B, Cahoon DR. 2002. Responses of coastal wetlands to rising sea level. *Ecology* **83**: 2869–77.

Morris B, Turner I. 2010. Morphodynamics of intermittently open-closed coastal lagoon entrances: new insights and a conceptual model. *Marine Geology* **271**: 55–66.

Möller I. 2006. Quantifying saltmarsh vegetation and its effect on wave height dissipation: Results from a UK East coast saltmarsh. *Estuarine, Coastal and Shelf Science* **69**: 337–51.

Morzaria-Luna HN and Zedler JB. 2014. Competitive Interactions Between Two Salt Marsh Halophytes Across Stress Gradients. *Wetlands* **34**: 31–42.

Mudd SM, D'Alpaos A, Morris JT. 2010. How does vegetation affect sedimentation on tidal marshes? Investigating particle capture and hydrodynamic controls on biologically mediated sedimentation. *Journal of Geophysical Research* **115**: F03029.

NOAA National Ocean Service Office for Coastal Management. 2012. *2009 – 2011 CA Coastal Conservancy Coastal Lidar Project: Hydro-flattened Bare Earth DEM*. NOAA Office for Coastal Management, Charleston, SC, USA. Viewed March 2019.

Oppenheimer, M., B.C. Glavovic , J. Hinkel, R. van de Wal, A.K. Magnan, A. Abd-Elgawad, R. Cai, M. Cifuentes-Jara, R.M. DeConto, T. Ghosh, J. Hay, F. Isla, B. Marzeion, B. Meyssignac, and Z. Sebesvari, 2019: Sea Level Rise and Implications for Low-Lying Islands, Coasts and Communities Supplementary Material. In: IPCC Special Report on the Ocean and Cryosphere in a Changing Climate [H.-O. Pörtner, D.C. Roberts, V. Masson-Delmotte, P. Zhai, M. Tignor, E. Poloczanska, K. Mintenbeck, A. Alegría, M. Nicolai, A. Okem, J. Petzold, B. Rama, N.M. Weyer (eds.)]. In press.

Orescanin MM, Scooler J. 2018. Observations of episodic breaching and closure at an ephemeral river. *Continental Shelf Research* **166**: 77-82.

Ortega-Cisneros K, Scharler U, Whitfield A. 2014. Inlet mouth phase influences density, variability and standing stocks of plankton assemblages in temporarily open/closed estuaries. *Estuarine, Coastal and Shelf Science* **136**: 139-148.

Page HM, Petty RL, and Meade DE. 1995. Influence of watershed runoff on nutrient dynamics in a Southern California salt marsh. *Estuarine, Coastal and Shelf Science* **41**: 163–80.

Pezeshki S.R. 2001. Wetland plant responses to soil flooding. *Environmental and Experimental Botany* **46**: 299-312.

Pérez-Ruzafa A, Marcos C, Pérez-Ruzafa I. 2011. Mediterranean coastal lagoons in an ecosystem and aquatic resources management context. *Physics and Chemistry of the Earth, Parts A/B/C* **36**: 160–166.

Polade SD, Gershunov A, Cayan DR, *et al.* 2017. Precipitation in a warming world: Assessing projected hydro-climate changes in California and other Mediterranean climate regions. *Scientific Reports* **7**.

Pollard D. 1994. A comparison of fish assemblages and fisheries in intermittently open and permanently open coastal lagoons on the south coast of New South Wales, south-eastern Australia. *Estuaries* **17**: 631.

R Core Team. 2016. R: a language and environment for statistical computing. Vienna, Austria.

Reed DJ. 1995. The response of coastal marshes to sea-level rise: Survival or submergence? *Earth Surface Processes and Landforms* **20**: 39–48.

Rich A, Keller EA. 2013. A hydrologic and geomorphic model of estuary breaching and closure. *Geomorphology* **191**: 64–74.

Rojas O, Mardones M, Martínez C, Flores L, Sáez K, Araneda A. 2018. Flooding in Central Chile: Implications of Tides and Sea Level Increase in the 21stCentury. *Sustainability* **10**(12): 4335.

- Rosencranz, J.A., N.K. Ganju, R.F. Ambrose. S.M. Brosnahan, P.J. Dickhudt, G.M. MacDonald, J.Y. Takekawa, and **K.M. Thorne**. *2015*. Balanced sediment flux in southern California's Mediterranean-climate salt marshes. *Estuaries and Coasts*, DOI 10.1007/s12237-015-0056-y

Schallenberg M, Larned ST, Hayward S, Arbuckle C. 2010. Contrasting effects of managed opening regimes on water quality in two intermittently closed and open coastal lakes. *Estuarine, Coastal and Shelf Science* **86**(4): 587-597.

Schile L, Callaway J, Morris J, Stralberg D, Parker VT, Kelly M. 2014. Modeling tidal marsh distribution with sea-level rise: evaluating the role of vegetation, sediment, and upland habitat in marsh resiliency. *PloS one* **9**(2): e88760.

Stretch D, Parkinson M. 2006. The breaching of sand barriers at perched, temporary open/closed estuaries – A model study. *Coastal Engineering Journal* **48**(1): 13-30.

Swain DL, Langenbrunner B, Neelin JD, and Hall A. 2018. Increasing precipitation volatility in twenty-first-century California. *Nature Climate Change* **8**: 427.

Swanson KM, Drexler JZ, Schoellhamer DH, Thorne KM, Casazza ML, Overton CT, Callaway JC, Takekawa JY. 2014. Wetland accretion rate model of ecosystem resilience (WARMER) and its application to habitat sustainability for endangered species in the San Francisco Estuary. *Estuaries and Coasts* **37**: 476–492.

Thorne K, MacDonald G, Guntenspergen G, Ambrose R, Buffington K, Dugger B, Freeman C, Janousek C, Brown L, Rosencranz J, Holmquist J. 2018. US Pacific coastal wetland resilience and vulnerability to sea-level rise. *Science Advances* **4**(2): eaao3270.

Tran T, Nguyen XH, Huynh TLH, Tran VT, Duong NT, Doan TTH. 2017. Estimating sea level rise for Vietnam East Sea. *Vietnam Journal of Science, Technology and Engineering* **59**(1): 73-78.

White NJ, Haigh ID, Church JA, Koen T, Watson CS, Pritchard TR, Watson PJ, Burgette RJ, McInnes KL, You ZJ, Zhang X. 2014. Australian sea levels—Trends, regional variability and influencing factors. *Earth-Science Reviews*, **136**: 155-174.

Whitfield AK, Bate G, Adams J, Cowley PD, Froneman PW, Gama PT, Strydom NA, Taljaard S, Theron AK, Turpie JK, Van Niekerk L. 2012. A review of the ecology and management of temporarily open/closed estuaries in South Africa, with particular emphasis on river flow and mouth state as primary drivers of these systems. *African Journal of Marine Science* **34**: 163–180.

Whitfield AK, Elliott M. 2012. Ecosystem and biotic classifications of estuaries and coasts. In: *Treatise on Estuaries and Coasts*. Elsevier, Amsterdam.

Williams G, Noe G, and Desmond J. 1998. The physical, chemical, and biological monitoring of Los Peñasquitos Lagoon, 1997-98. *Final report prepared for the Los Peñasquitos Lagoon Foundation*.

Winter SA, 2020. An Investigation of the Closure Regimes of California's Bar-Built Estuaries MS Thesis, University of California Davis, 83pp.

Wooldridge T.H. 1994. The effect of periodic inlet closure on recruitment in the estuarine mudprawn. In: *Changes in Fluxes in Estuaries*: 329-333. Olsen & Olsen, Fredensborg, Denmark.

Young GC, Potter IC, Hyndes GA, de Lestag S. 1997. The ichthyofauna of an intermittently open estuary: Implications of bar breaching and low salinities on faunal composition. *Estuarine, Coastal and Shelf Science* **45**(1): 53-68.

Zedler JB. 1996. Coastal mitigation in southern California: The need for a regional restoration strategy. *Ecological Applications* **6**:84-93.

Zedler JB, Covin J, Nordby C, *et al.* 1986. Catastrophic events reveal the dynamic nature of salt-marsh vegetation in southern California. *Estuaries and Coasts* **9**: 75–80.