2019

Early results – salt marsh response to changing fine sediment supply conditions, Humboldt Bay, CA

USGS

Early results - salt marsh response to changing fine-sediment supply conditions, Humboldt Bay, CA

Jennifer A. Curtis, Geologist, U.S. Geological Survey, Eureka, CA, jacurtis@usgs.gov
Chase Freeman, Biologist, U.S. Geological Survey, Davis, CA, cfreeman@usgs.gov
Karen Thorne, Research Ecologist, U.S. Geological Survey, Davis, CA, kthorne@usgs.gov

1.0 Introduction

The resiliency and vulnerability of natural and restored salt marshes is highly dependent upon the mineral sediment supply (Weston, 2014; Ganju et al.,2015) carried by the water that inundates the marsh surface. Marsh surface elevations are maintained through complex morpho-dynamics and marsh evolution models assume that sediment deposition, vertical accretion and elevation gain are directly proportional to suspended-sediment concentrations (Kirwan and Murray, 2007; Fagherazzi et al., 2012). In this study we use direct measurements of vertical accretion, marsh elevation change, and suspended-sediment concentrations (SSC) to investigate salt marsh response to changing fine-sediment (<63 µm) supply conditions in Humboldt Bay, CA.

Both mineral- and organic-sediment supply maintain marsh surface elevations (D'Alpaos et al., 2011; Thorne et al., 2016), which must keep pace with relative sea-level rise (RSLR) to avoid submergence and conversion to subtidal habitat if marsh transgression is not possible (Kirwan et al., 2010; Thorne et al., 2018). Modeling and field-based studies agree that sediment-rich marshes are less vulnerable to RSLR and sediment-limited marshes are more vulnerable to RSLR (Patrick and DeLaune, 1990; Thom, 1992; Stralberg et al., 2011; Thorne et al., 2016).

There is a dynamic balance that exists between the rates of RSLR, local morphology, sediment supply, hydrodynamics, plant productivity, and the ability of marsh vegetation to trap and stabilize available sediment (Thom, 1992; Callaway et al, 1996; Cahoon, 1997; Morris et al., 2002). To manage and restore salt marshes effectively and sustainably, we need to understand resiliency and how they respond to changing sediment supply conditions. In Humboldt Bay, where long-term RSLR ranges from 3.11 to 5.56 mm/yr (Anderson, 2015), which is greater than most west coast regions due to tectonic subsidence (Russell, 2012; Montillet et al., 2018), an adequate sediment supply is critical if existing and restored salt marshes are to persist into the future. This study was designed to inform management actions that may affect the trajectory of vertical marsh accretion and vulnerability to sea-level rise (SLR) such as regional sediment management, dredging, and tidal restoration to subsided former baylands.

2.0 Regional Setting

Humboldt Bay is located on the north coast of California (Figure 1). The bay is protected by coastal barriers and sand spits but is subject to energetic conditions driven by storms, waves, and wind events. Costa (1982) described the bay as a tide-driven coastal lagoon with limited freshwater contributions that occur primarily during large winter storms. There are three subembayments referred to as the Entrance Bay, North Bay and South Bay. The subembayments are connected by the entrance channel and a network of navigation channels that require periodic maintenance dredging (HBHRCD, 2007). Dredging began in 1881 and currently the average annual volume of dredged fine-sediment (<63 µm) is approximately 60,500 m3 (CCSMW, 2017), which equates to 0.10 Mt/yr using a conversion factor of 1.7 Mt/m3.

The sheltering effect of the barrier spits protects the interior of the bay from wave exposure and allowed expansive areas of salt marsh to form historically in low energy

environments along the bay margins. In 1870 salt marshes occupied approximately 36 km2 (Figure 1) but the present distribution represents less than 10% of the former extent (Pickart, 2001). Currently, salt marshes exist as fragments along the bay's margins, at the mouths of local tributaries, or recessed upstream within tidal slough channels. Approximately 70, 25, and 5% of the remaining salt marshes (<3.6 km2) are found in the North Bay, Entrance Bay and South Bay, respectively (Schlosser and Eicher, 2012). These tidal marshes are important habitat for migratory and resident birds and juvenile coho salmon (Oncorhynchus kisutch).

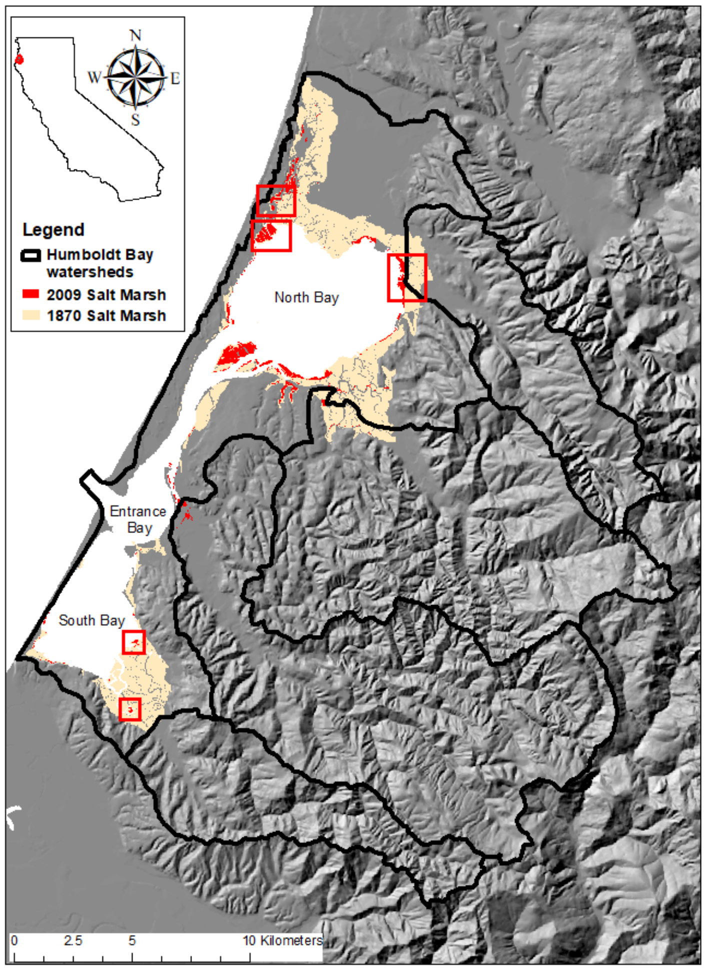

Figure 1. Humboldt Bay study area showing spatial extent of tidal salt marshes in 1870 (Laird, 2007) and 2009 (Schlosser and Eicher, 2012). Red bounding boxes delineate five salt marsh study sites (see Figure 2 for detailed study marsh maps).

2.1 Hydrodynamics

Humboldt Bay is relatively shallow with 39 km2 of mudflats exposed at mean lower low water (MLLW) and the mean daily tidal exchange volume is approximately 114 million m3/day (Anderson, 2015). The exchange volume, or tidal prism, is quite large in comparison to the freshwater discharge from the local watersheds. The mean annual freshwater discharge is

approximately 0.6 million m3/yr (Curtis et al., in review). The relatively small freshwater inflow from the bay watersheds results in tidally-dominated circulation, with estuarine conditions existing only during the winter-runoff season at the tributary-bay interface.

The bay experiences mixed-semidiurnal tides with a mean diurnal range of 2.1 meters (estimated as the difference between MLLW and MHHW) and mean tide of 1.49 meters (National Oceanic and Atmospheric Agency Station, North Spit, 9418767; https://tidesandcurrents.noaa.gov/). The North Bay is deeper relative to South Bay and the contributions to the tidal prism are ~50% and ~25% respectively (Anderson, 2015).

Notably, the flushing rates of North Bay are lower than South Bay due to the bay's morphology (Costa, 1982) and this influences the amount of marine-derived sediment that can enter and the amount of freshwater-derived sediment that can exit. Because the volume of the three subembayments is large in comparison to the tidal channels, water that flows into the bay on a high tide cannot be completely replaced during a single tidal exchange. Approximately 41% of the water is replaced during each tide cycle and full tidal exchange can take 4 to 21 days (Schlosser and Eicher, 2012).

2.2 Climate, hydrology, and fine-sediment supply

Humboldt Bay is located at the transition between the Pacific Northwest and California climate regions, within the Coast Range geologic province, and has a Mediterranean climate with distinct cool-dry summers and mild-wet winters. The average annual precipitation is 1,585 mm/yr, of which only 3% falls between June and September (Curtis et al., in review). The orographic effect of the Coast Range creates a strong precipitation gradient and the hydrology is characterized by extremes. Winter discharge peaks are typically rainfall-driven, and snowmelt plays a less significant role. However heavy rain events, referred to as atmospheric rivers (Dettinger et al., 2011), can produce dramatic floods (Brown and Ritter, 1971; Waananen, 1971).

Watersheds that deliver sediment to the north coast of California are characterized by steep-forested uplands and low-lying areas near the mouth composed of floodplains, pastures and wetlands. These coastal watersheds have high rates of fine-sediment yield related to regional tectonics, erodible lithology, climate and land use history (Brown and Ritter, 1971; Kelsey, 1980; Milliman and Farnsworth. 2001; Warrick et al., 2013).

Humboldt Bay receives direct inputs of fine-sediment and freshwater from several small tributary watersheds with a combined contributing area of 442 km2 (Figure 1). Historically, the upland forests were extensively logged (Leithold et al., 2005; Klein et al., 2012) and low-lying areas have been diked and leveed (Schlosser and Eicher, 2012).

The coastal sediment budget is dominated by sediment discharged from the Eel River (9,415 km2) during winter runoff events (Wheatcroft et al., 1997; Wheatcroft and Borgeld, 2000; Farnsworth and Warrick, 2007, Warrick, 2014). Sediment discharge from the coastal rivers of northern California peaked in water year 1965 and have since declined (Warrick et al., 2013). The peak in sediment discharge was related to intense logging and a devastating flood in 1964 (Brown and Ritter, 1971; Waananen, 1971). Because the daily tidal exchange within Humboldt Bay is much larger than the annual freshwater input, the bay may be a sink for fine-sediment derived from oceanic sources but there are no direct measurements available to support this assertion.

3.0 Salt Marsh Descriptions

We selected five study marshes (Table 1) distributed throughout Humboldt Bay (Figure 2) for monitoring salt marsh accretion and elevation change. Two of the sites (Mad River and Manila) were established in 2013. Baseline measurements for this study began in November of 2015. Mad River marsh and Manila marsh are in the western region of North Bay. Mad River

marsh is a high elevation island marsh located upstream within Mad River Slough; while Manila marsh is a low elevation fringe marsh located at the bay margin. Sediment is supplied from the tidal channels; however, there is freshwater drainage from the dunes to the west and a perennial stream that emerges at the base of the moving dunes that discharges to Mad River Slough. Jacoby marsh, located on the eastern edge of North Bay at the mouth of Jacoby Creek, is a high elevation deltaic marsh with direct inputs of freshwater and sediment. White marsh and Hookton marsh are in the eastern region of South Bay. White marsh is a low elevation island marsh located at the bay margin; while Hookton marsh is a low elevation island marsh located upstream within Hookton Slough. Salmon Creek flows into Hookton Slough downstream from Hookton marsh and supplies direct inputs of freshwater and sediment.

Four of the study marshes (Mad River, Manila, White and Hookton) are within the USFWS Humboldt Bay National Wildlife Refuge and are part of a regional Spartina densiflora eradication program. *S.densiflora* is an invasive cordgrass that has infested approximately 90% of the salt marshes within Humboldt Bay (Pickart, 2001). Manila marsh, managed by the California Department of Fish and Wildlife, is not part of the eradication program. In 2006, pilot studies for mechanical treatments to remove *S.densiflora* began in Mad River Slough and in 2010 a regional eradication effort began (Pickart, 2012). During mechanical treatments low elevation zones and microtopography are created that could contribute to incremental lowering of marsh surface elevations. Pickart (2013) conducted repeat laser level surveys at Jacoby marsh to measure changes in mean marsh elevations related to various *S.densiflora* treatments. After 1.5 years marsh elevations had recovered and were within +/-1.3 cm of the baseline elevations; but this may have been accelerated due to the site being located at the mouth of Jacoby Creek, which is one of the primary tributaries that contributes sediment to the bay (Curtis et al., in review).

Table 1. Descriptions and attribute information for five salt marshes located in Humboldt Bay, CA. Relative sea-level rise (RSLR) estimates are from Anderson, 2015.

Site Name	Geomorphic Setting	Area (km^2)	RTK-GPS (Number of points)	Elevation (NAVD88) Mean (m)	Elevation (NAVD88) Range (m)	Spartina Treatment	Base Line Date	RSLR (mm/yr)
North Bay Marshes								
Mad River	Island	0.06	852	2.05	1.20-2.29	2006, 2008, 2013 + maintenance	11/19/15	3.11
Manila	Fringe	0.13	732	1.72	0.79-2.53	none	11/19/15	3.11
Jacoby	Deltaic	0.12	558	2.02	1.03-2.43	2010, 2011 + maintenance	11/20/15	3.11
South Bay Marshes								
White	Island	0.03	109	1.79	1.00-1.99	2010, 2011+ maintenance	11/22/15	5.56
Hookton	Island	0.02	83	1.83	1.12-2.17	2010, 2011+ maintenance	11/22/15	5.56

4.0 Methods

4.1 Marsh Elevation and Vertical Accretion Monitoring

We installed deep rod Surface Elevation Table (SET) and feldspar marker horizon (MH) plots (Figure 3) to quantify the relative contributions of surface and subsurface processes to vertical accretion and elevation change in each of the five study marshes. The SET-MHs were installed in Mad River and Manila marshes in 2013. The SET-MHs were installed in Jacoby, White and Hookton marshes in 2015. A summary of the SET-MH protocol was published by Lynch et al. (2015).

Vertical changes in the marsh surface are the result of accretion, erosion, decomposition, compaction, shrink-swell caused by groundwater flux, swell caused by root growth, and deeper processes such as regional subsidence or uplift. The SET measurements quantify surface elevation change and the MH measurements quantify vertical accretion above a feldspar layer applied on the marsh surface. Vertical accretion is defined as the buildup of mineral and organic sediment on the marsh surface, and elevation change is defined as a change in the height of the wetland surface relative to a local benchmark.

At each study marsh two representative sites were selected after considering surface elevations, vegetation composition and distance from tidal sources (Figure 2). One SET and three MHs were deployed at each site (a total of two SETs and six MHs per marsh) following standardized methods (Cahoon et al., 2002; Webb at al., 2013). SET-MHs were measured during quarterly site visits. Measurement of the MH entails removing a small plug of soil using a soil knife, measuring the depth of surface accretion above the feldspar layer, and replacing the plug. Elevation change is measured by attaching the SET instrument to a collar installed at the top of the local benchmark, in this case the top of the deep rod. The SET instrument provides a constant reference plane in space from which the distance to the marsh surface can be measured. Nine pins are lowered to the surface in four ninety-degree cardinal directions yielding 36 observations. Repeat measurements can resolve millimeter-scale change (Cahoon et al, 2002) because the orientation of the table in space remains fixed in time.

4.2 Bias-Corrected Digital Elevation Model Generation

Baseline elevation RTK-GPS surveys, completed in 2012 and 2013 at the five study marshes (Takekawa et al., 2013), were used to correct the vegetation bias in an available bare-earth high resolution (1 meter) digital elevation model (DEM; CA-SCC, 2012). The bias-corrected DEM was used to estimate the marsh elevations presented in Table 1. Elevations were surveyed using a Leica survey-grade GNSS rover (Viva GS15 and RX1250X models). GPS real-time kinematic (RTK) corrections were streamed to the rover from a Leica base station (Leica GNSS Receiver GS10 with Leica AS10 antenna) during the surveys. The mean vertical error was ±2 cm (Thorne et al. 2015, Thorne et al., 2016) and the ellipsoid heights of the marsh surface were post-processed to determine orthometric heights referenced to NAVD88 and the geoid 12A model.

The RTK-GPS elevations and a Normalized Difference Vegetation Index (NDVI) were used to correct a positive bias in the marsh DEMs related vegetation cover using the LEAN method (Buffington et al., 2016). We obtained LiDAR-derived DEMs from the Digital Coastal Data Access Viewer (https://coast.noaa.gov/dataviewer/) and 2016 multispectral airborne imagery data from the National Agriculture Imagery Program (NAIP; https://catalog.data.gov/dataset/naip-public-image-services). From the NAIP imagery we calculated an NDVI:

$$NDVI = ([NIR-Red]/[NIR+Red])$$

where “Red” included wavelengths of 608-662 nm and “NIR” included wavelengths of 833-887 nm. Using the LEAN method, the positive bias in the LiDAR -DEM was calculated by determining elevations difference between the LiDAR -DEM and the RTK-GPS elevations. We then used a multivariate linear regression model to define a statistical relationship between LiDAR error, NDVI, and LiDAR elevation. The regression model was used to develop bias-corrected mean elevations estimates for each study marsh (Table 1).

4.3 Water Quality and Suspended-Sediment Monitoring

Water quality stations (Table 2) were established in Mad River Slough (USGS 405219124085601 MAD R SLOUGH NR ARCATA CA) and Hookton Slough (USGS 404038124131801 HOOKTON SLOUGH NR LOLETA C) in the primary tide tidal channels that supply sediment to the adjacent study marshes (Figure 2). Water quality sondes (YSI-EXO2), equipped with a turbidity sensor and a combined temperature and specific conductance sensor, were deployed in March of 2016 at a fixed water depth of 1.0 meter. The sondes and sensors were cleaned monthly and calibrations checked during quarterly site visits. Specific conductance was converted to salinity using a temperature (25^{o} C) compensated method (*Wagner et al., 2006*). Continuous 15-minute records of turbidity, temperature, specific conductance, and salinity are available for each station at https://waterdata.usgs.gov/nwis/qw .

Water samples were collected, and water depths measured, during quarterly site visits. A Van-Dorn sampler was used to collect 1-liter water samples throughout a rising and falling tide at 1.5-hour intervals. During each visit one replicate sample was collected to address variability and field blanks were collected periodically to verify adequate cleaning procedures. Water samples were stored in brown HDPE bottles, kept cool and shipped to the USGS Cascade Volcanic Observatory sediment laboratory (Vancouver, WA) for analysis. Suspended-sediment concentrations (SSC) were determined by filtration methods for all the samples. Due to funding limitations percent organic material was determined by loss on ignition (LOI) for a subset of samples, typically two samples per site per visit. The water sample data are also available for each station at https://waterdata.usgs.gov/nwis/qw.

Turbidity can be used as a surrogate for SSC (*Rasmussen et al., 2009*) and we used ordinary least-squares regression to convert the turbidity time series to SSC. The time and date stamp for each of the water samples was synced with the turbidity time series to determine associated turbidity values. A least-squares linear regression equation was determined using the lab-derived SSC and associated turbidity values. The regression model was used to convert turbidity values to SSC and derive a continuous 15-minute SSC time series. The converted SSC time series was used to assess variations in SSC and to investigate correlations with marsh accretion measurements.

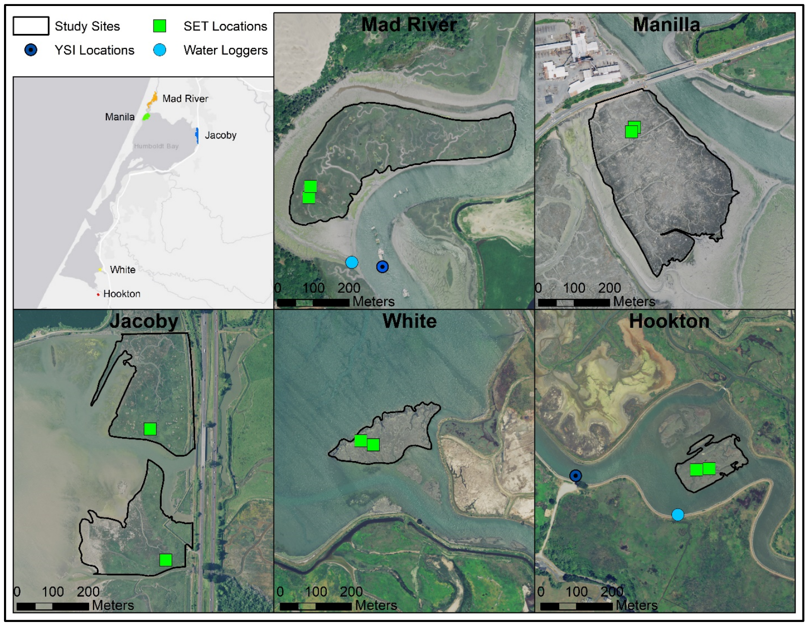

Figure 2. Five study marsh monitoring sites in Humboldt Bay, CA. Map shows the location of study marshes, Sediment Elevation Tables (SET), Marker Horizons (MH), water quality sondes (YSI), and water level loggers.

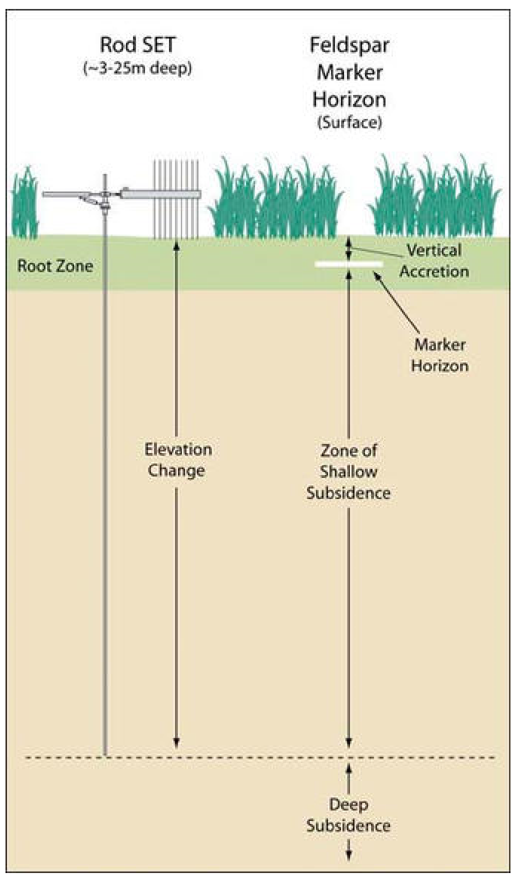

Figure 3. Conceptual diagram showing how the soil profile is measured to assess marsh surface and subsurface processes by Surface Elevation Table (SET) and Marker Horizon (MH) techniques (Cahoon et al, 2002).

Table 2. Descriptions of two water quality monitoring stations located in Humboldt Bay, CA.

Water Quality Station	Instruments	Parameters	Easting	Northing	Deployment Date
USGS 405219124085601 MAD R SLOUGH NR ARCATA CA	YSI-EXO2	Turbidity (FNU) Specific conductance (µs/cm @25°C) Temperature (°C)	403198	4525162	3/5/2016 - present
	Hobo U20	Water level (m)	403133	4525173	3/17/2016 – 12/8/16
	LT Edge (2…)	Water level (m)	403133	4525173	12/8/16 - present
USGS 404038124131801 HOOKTON SLOUGH NR LOLETA C	YSI-EXO2	Turbidity (FNU) Specific conductance (µs/cm @25°C) Temperature (°C)	396746	4503666	3/5/2016 - present
	Hobo U20	Water level (m)	397033	4503557	3/17/2016 – 12/8/16
	LT Edge (2…)	Water level (m)	397033	4503557	12/8/16 - present

5.0 Results

5.1 Marsh Elevation and Accretion Measurements

There were nine SET-MH measurements collected during the 2-year study period between November 22nd, 2015 and December 3rd, 2017. Again, SET measurements quantify elevation change and feldspar MH measurements quantify vertical accretion (Cahoon et al., 2002; Lynch et al., 2015). If vertical accretion is greater than elevation change, shallow subsidence (accretion minus elevation change) related to decomposition or compaction may be occurring. If accretion is equal to elevation change we can infer that surface accretion is driving elevation change and subsurface processes are negligible. If accretion is less than elevation change we can infer that shallow expansion related to swelling of soils by water storage or an increase in root volume may be occurring.

Over the 2-year study period elevation changes and accretion was spatially and temporally variable (Table 3). At the South Bay sites (Hookton and White) accretion rates were about 1.5 times greater than elevation changes; but changes in elevation and accretion were about equal at the North Bay sites (Mad River, Manila, and Jacoby). Across all the sites elevation change and accretion were lower during 2016 (-0.26mm ±0.64; 1.56mm ±1.66) and higher in 2017 (3.15mm ±0.30; 2.82mm ±1.04).

We also compared the annual rates of elevation change and accretion to estimates of long-term trends in RLSR (Figure 4) estimated for the Humboldt Bay region (Anderson, 2015). RSLR estimates for North Bay and South Bay are 3.11 mm/yr and 5.56 mm/yr respectively (Table 1). During the 2-year study period the rates of annual elevation gain did not outpace long-term trends in RSLR; however, these short-term results represent initial baseline measurements and should be interpreted with caution within the framework of the longer-term trends in RSLR. Continued monitoring, over decadal or longer periods, is required to detect trends in elevation gain and vertical accretion.

Table 3. Summary of elevation change and accretion measurements and the associated standard errors over a 2-year period for five study marshes located in Humboldt Bay, CA.

	2016		2017		Cumulative		Average Annual	
Site	**Elevation change (mm)**	**Accretion (mm)**	**Elevation change (mm)**	**Accretion (mm)**	**Elevation change (mm)**	**Accretion (mm)**	**Elevation change (mm/yr)**	**Accretion (mm/yr)**
North Bay Marshes								
Mad River	-0.89 ± 0.37	-4.29 ± 0.21	0.52 ± 0.18	5.33 ± 0.46	-0.38 ± 0.55	1.04 ± 0.67	-0.19 ± 0.28	0.52 ± 0.34
Manila	-3.04 ± 1.54	-0.3 ± 4.67	5.54 ± 0.44	0.67 ± 2.08	2.50 ± 1.98	0.36 ± 6.75	1.25 ± 0.99	0.19 ± 3.38
Jacoby	0.71 ± 0.40	2.13 ± 0.88	2.49 ± 0.18	0.75 ± 0.04	3.19 ± 0.58	2.88 ± 0.92	1.60 ± 0.29	1.44 ± 0.46
South Bay Marshes								
Hookton	1.09 ± 0.52	7.90 ± 2.23	3.25 ± 0.60	1.60 ± 2.08	4.34 ± 1.12	9.50 ±4.31	2.17 ± 0.56	4.75 ± 2.16
White	0.81 ± 0.37	2.38 ± 0.38	3.95 ± 0.11	5.75 ± 0.54	4.76 ± 0.48	8.13 ± 0.92	2.38 ± 0.24	4.07 ± 0.46
North Bay	-1.07 ± 0.77	-0.82 ± 1.89	2.85 ± 0.27	2.25 ± 0.86	1.77 ± 1.04	1.43 ± 2.78	0.89 ± 0.52	0.71 ± 1.39
South Bay	0.95 ± 0.45	5.14 ± 1.31	3.60 ± 0.36	3.68 ± 1.31	4.55 ± 0.80	8.82 ± 2.62	2.28 ± 0.40	4.41 ± 1.31
All sites	-0.26 ± 0.64	1.56 ± 1.66	3.15 ± 0.30	2.82 ± 1.04	2.88 ± 0.94	4.38 ± 2.71	1.44 ± 0.47	2.19 ± 1.36

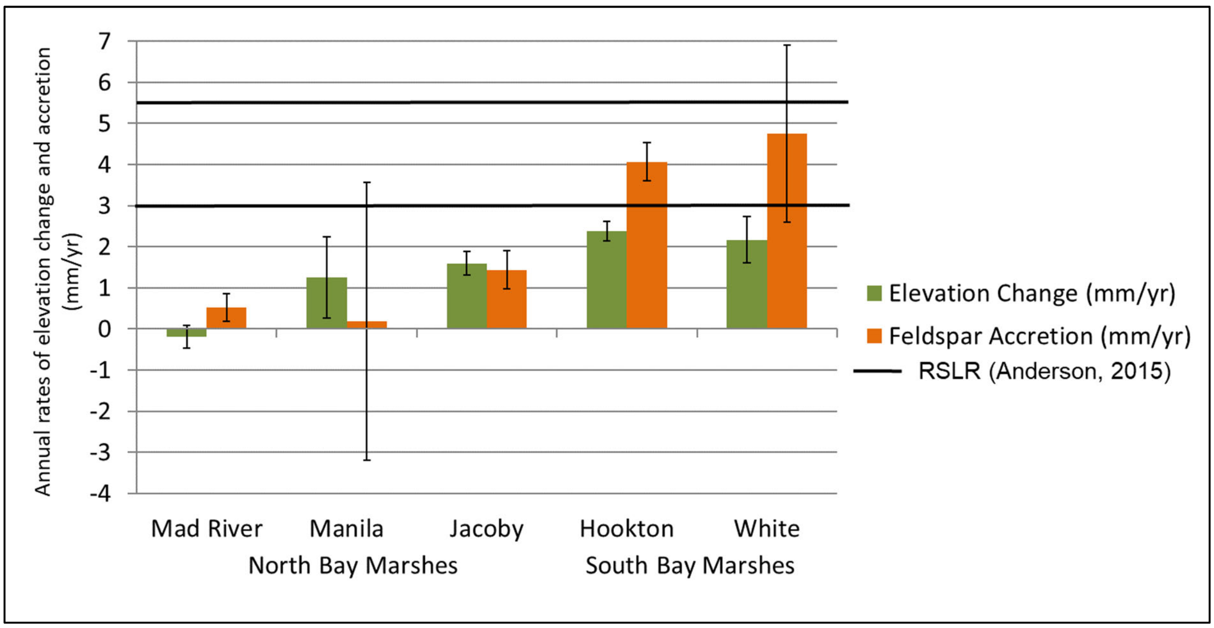

Figure 4. Summary of mean annual rates of elevation change and accretion for five study marshes located in Humboldt Bay, CA. When accretion is greater than elevation change this indicates shallow subsidence that can be caused by decomposition and compaction. When elevation change is greater than accretion this indicates accumulation of below-ground biomass or swelling of soils by water storage. The range of relative sea level rise (RSLR; Anderson, 2015) for Humboldt Bay (3.11 to 5.56 mm/yr) is shown with horizontal black lines. Uncertainty in the elevation change and accretion measurements is captured by the standard error shown as vertical error bars.

5.2 Water Quality and Suspended-Sediment Supply

We converted the turbidity records into a SSC time series using eq.1 and eq.2 and computed summary statistics for each monitoring station (Table 4). The mean SSC measured at Hookton slough (41.1 mg/L) was 2.5 times greater than the mean SSC measured at Mad River slough (16.8 mg/L). The median SSC values for the two sites were similar indicating that the bay

is well-mixed and tidally-dominated for most of the year. The standard deviation (SD), coefficient of variation (CV) and range in SSC values are measures of statistical variance, which were much greater at Hookton indicating more variability in the sediment supply due to large episodic freshwater inputs.

Hookton SSC = 1.274+ 1.95 * Turbidity r^2= 0.928 $p < 0.0001$, N=46 eq.1
Mad River SSC = 4.14+ 1.26 * Turbidity r^2= 0.396 $p < 0.0001$, N=45 eq.2

The lack of variance in SSC measurements at Mad River Slough heavily influenced the regression model used to convert the turbidity signal to SSC values. Although the p-values indicate the Mad River and Hookton regression models are statistically significant, the lack of variance in the SSC values for the Mad River model resulted in a much lower slope and r^2 value.

Table 4. Statistical metrics for suspended-sediment concentrations (SSC) derived from continuous turbidity records collected over a 2-year study period at two water quality monitoring stations in Humboldt Bay, CA. Note: SD is the standard deviation and CV is the percent coefficient of variation.

Monitoring Station Location	USGS Water Quality Station Number	Mean SSC (mg/L)	SD SSC (mg/L)	CV SSC (%)	Min SSC (mg/L)	Max SSC (mg/L)	Median SSC (mg/L)
Mad River Slough	405219124085601	16.8	7.1	42	4.9	414.0	15.7
Hookton Slough	404038124131801	41.1	81.5	198	8.0	1598.0	19.7

6.0 Discussion

6.1 Geomorphic stability and vulnerability to SLR

Sediment supply is a primary variable for determining geomorphic stability and salt marsh vulnerability to RSLR (*Callaway, 1996; Pethick and Crooks 2000; Weston, 2014; Ganju et al., 2015 Thorne al., 2016*). Sufficient sediment supply must be available for salt marshes to gain elevation and persist in place. This study focused on direct measurements of three variables that control salt marsh resiliency and vulnerability to SLR in Humboldt Bay: fine-sediment supply, marsh elevation, and marsh accretion.

Salt marshes respond dynamically to accommodate change and have been referred to as “ephemeral landforms” (*Orr et al., 2003*). In general, wave and tidal energy is attenuated through the transfer of sediment from high-energy source areas, where transport and erosion occur, to low-energy sinks where sediment deposition and accumulation occurs. This transfer of sediment and the associated energy attenuation creates a strong morpho-dynamic response with wave and tidal energy creating morphologic change, which creates feedback that alters the local energy environment (*Pethick 1996; D’Alpaos et al., 2011; Fagherazzi et al., 2012*). The form and function of salt marshes therefore depends upon a dynamic balance between the energy regime and the transport and deposition of fine-sediment.

During periods of increased coastal energy, the natural marsh response is landward transgression to lower energy environments while the seaward edge of the marsh experiences erosion and is replaced by mudflat and subtidal habitat. Approximately 75% of the bay’s shoreline is composed of artificial hard structures, including Highway 101 and a former railroad grade (*Laird, 2013*). Under current conditions much of the space to accommodate dynamic marsh transgression has been lost.

Recent studies indicate that sediment transport-based metrics are good indicators of vulnerability and wetland stability (*Ganju et al., 2013; Ganju et al., 2015*). In this study, we

assumed that SSC is representative of the fine-sediment supply available for accretion. The SSC-metrics in Table 4 indicate Hookton slough (South Bay) is sediment-rich. In comparison, Mad River slough (North Bay) is sediment-limited with less fine-sediment available for accretion.

We further investigated our results to assess the correlation between accretion and SSC in Hookton and Mad River marsh using quarterly measurements of accretion and the average SSC estimated during eight quarterly intervals over the 2-year study period. There was a positive correlation between accretion and SSC for Hookton marsh where sediment-rich conditions exist, but the correlation was not statistically significant (r^2=0.16, p=0.3270). There was no correlation for Mad River marsh, where sediment-limited conditions exist ($r^2$0.00, p=0.9025) (Figure 5). Additional data collected in 2018 and 2019 may improve the correlations

In summary, the North Bay is sediment-limited and is experiencing lower long-term rates of RSLR (3.11 mm/yr). Our early results show that the North Bay marshes (Mad River, Manila, and Jacoby) are experiencing lower rates of vertical accretion (0.71±1.39 mm/yr) and elevation change (0.89±0.52 mm/yr) but there is high uncertainty associated with these measurements. In comparison, South Bay is sediment-rich and is experiencing higher long-term rates of RSLR (5.56 mm/yr) due to tectonic subsidence, which is mitigated somewhat by higher rates of accretion (4.41±1.31mm/yr) and elevation change (2.28±0.40 mm/yr). The South Bay accretion rates were greater than elevation changes, which may indicate that shallow subsidence, related to decomposition or compaction, could be a limiting factor influencing elevation gains.

The sediment-limited conditions in North Bay make Mad River and Manila marshes more vulnerable to accelerated RSLR, however, Jacoby marsh is a deltaic marsh located in the eastern region of North Bay with higher fetch and wind-wave exposure. Generally, deltaic marshes tend to have higher accretion rates (*Cahoon et al., 2006*) and the direct input of fine-sediment at Jacoby marsh may mitigate vulnerability, in this higher energy but more sediment-rich region of North Bay. South Bay marshes are more vulnerable than North Bay marshes to submergence due to higher rates of RLSR, but this is mitigated somewhat by greater sediment supply.

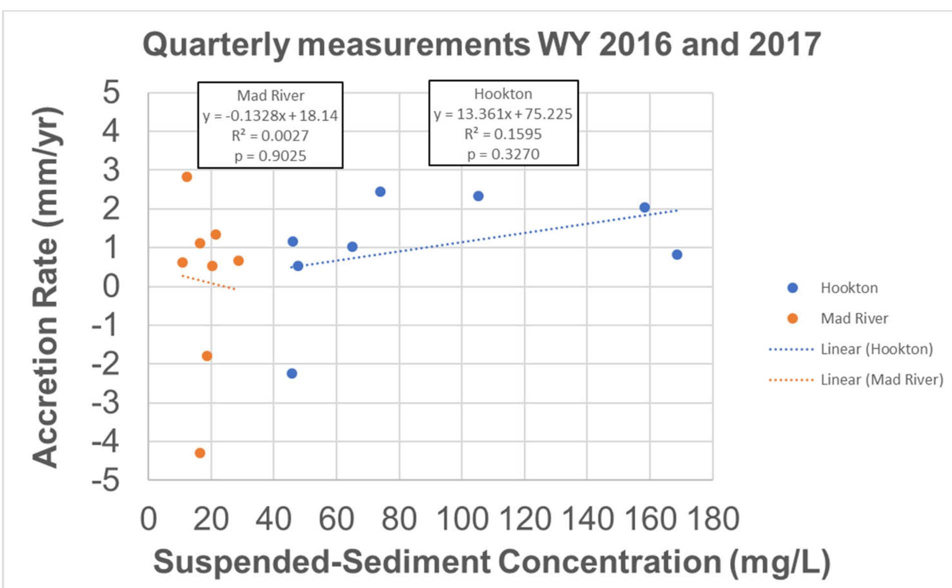

Figure 5. Correlation graph showing the relation between suspended-sediment concentration (SSC) and vertical accretion rates for two study marshes in Humboldt Bay, CA.

6.2 Fine-Sediment Budget and Management Implications

There are ongoing management and restoration activities that impact the fine-sediment budget of Humboldt Bay, which may alter the availability of sediment for marsh accretion and elevation gain. We assessed the potential impacts on the fine-sediment budget related to the regional *S.densiflora* eradication program, maintenance dredging of harbors and channels, and tidal restoration in subsided former baylands. All of these management activities alter local topography and create low elevation zones in the tidal prism. These low elevation zones impact the fine-sediment budget by increasing “sediment demand”, which may reduce the “sediment supply” available for marsh accretion and elevation gain.

The regional *S.densiflora* eradication program in Humboldt Bay uses mechanical treatments that create low-elevation microtopography. The impact of the *S.densiflora* treatments on marsh elevations was assessed at the Jacoby marsh (*Pickart, 2013*). Repeat laser level measurements indicated that after 1.5 years the surface elevations were within ± 1.3 cm of the original elevation. However, Jacoby marsh is a deltaic marsh with direct inputs of sediment and relatively high rates of accretion and elevation change and may not be representative of other North Bay marshes located in sediment-limited regions.

In a companion study Curtis et al. *(in review)* estimated the fine-sediment supply to Humboldt Bay from local watersheds (0.05 Mt/yr) and defined an imbalance created by maintenance dredging (0.10 Mt/yr). This fine-sediment deficit may be filled by natural deposition of sediment supplied from terrestrial or marine sources or by local recruitment of sediment within the bay through erosion of existing mudflats and marshes

Tidal restoration to subsided former baylands also impacts the fine-sediment budget by creating large “sediment sinks” and increasing “sediment demand”. There are several completed and planned tidal restoration projects within Humboldt Bay that involve strategically breaching dikes and levees to allow natural deposition and filling of subsided lands. A recently completed beneficial reuse study (*HBHRCD, 2015*) estimated the “sediment demand” associated with two projects in South Bay equates to 0.31 Mt, which is three times the annual maintenance dredging and 6 times the annual supply from the local watersheds.

Incorporating fine-sediment augmentation by direct placement into tidal restoration projects could ameliorate “sediment demand” and accelerate the rate of recovery to achieve adequate elevations to support salt marsh vegetation. A recent modeling study concluded that although RSLR is the primary controlling factor for marsh accretion and elevation gain, the starting surface elevation had the second greatest impact on elevation gain followed by the mineral-sediment supply (*Thorne et al., 2016*). Thus, initial elevation and sediment accretion rates, which are dependent on sediment supply, determine the effectiveness and success of salt marsh restoration.

Tidal restoration in subsided former baylands in sediment-rich areas of the bay may quickly fill and achieve the necessary elevations for the colonization of marsh vegetation. Conversely, projects located in sediment-limited areas may require augmentation to achieve desired increases in elevations to support marsh vegetation. Although sediment augmentation can add significantly to restoration project costs, and it may be a limiting factor, the beneficial reuse of dredged fine-sediment is one promising approach for salt marsh restoration that mitigates “sediment demand” and avoids recruitment of sediment from existing subtidal and intertidal habitats.

7.0 Conclusions and Future Work

This study improved our understanding of how salt marshes respond to changing sediment supply conditions in Humboldt Bay, CA. South Bay is shallower and rates of RSLR are

higher due to tectonic subsidence, but this is balanced by a larger sediment supply and higher rates of marsh accretion and elevation change. North Bay is deeper, much larger volumetrically with lower rates of RSLR, sediment supply, accretion, and elevation change. Salt marshes are highly dynamic systems that keep pace with SLR by vertical accretion and horizontal retreat when space for retreat is available. Without an adequate sediment supply, the salt marshes in Humboldt Bay are more vulnerable to submergence due to accelerated SLR. Early results indicate short-term rates of elevation gain were lower than the long-term estimates of RSLR for all five of the study marshes.

Continued monitoring of the fine-sediment budget, marsh accretion and elevation change is essential to understand the trajectory of marsh formation within the framework of accelerated SLR and to determine whether future management actions will be needed to mitigate additional marsh loss. With informed regional sediment management and environmental planning, it may be possible to mitigate the sediment demand created by management activities and associated impacts. Marsh augmentation, using excess fine-sediment derived from maintenance dredging, is a potential approach for alleviating imbalances in the fine-sediment budget that impact the sediment supply available for marsh accretion and elevation gain.

8.0 References

Anderson, J.K., 2015. Humboldt Bay: Sea Level Rise, Hydrodynamic Modeling and Vulnerability Mapping. Prepared for the State Coastal Conservancy and Coastal Ecosystems Institute of Northern California. McKinleyville, CA: Northern Hydrology & Engineering.

Brown III, W. M., and Ritter, J. R. 1971. Sediment transport and turbidity in the Eel River Basin, California. U.S. Geol. Survey. Water Supply Paper 1986.

Buffington, K.J., Dugger, B.D., Thorne, K.M. and Takekawa, J.Y., 2016. Statistical correction of lidar-derived digital elevation models with multispectral airborne imagery in tidal marshes. *Remote Sensing of Environment*, *186*, pp.616-625.

Cahoon, D. 1997. Global warming, sea-level rise, and coastal marsh survival. USGS National Wetlands Research Center. USGS FS-09 1-97.

Cahoon, D.R., Lynch, J.C., Hensel, P., Boumans, R., Perez, B.C., Segura, B., and Day, J.W., 2002, High precision measurements of wetland sediment elevation—I, recent improvements to the sedimentation-erosion table: Journal of Sedimentary Research, v. 72, p. 730–733.

Cahoon, D.R., Hensel, P.F., Spencer, T., Reed, D.J., Mckee, K.L., and Saintilan, N., 2006. Coastal wetland vulnerability to relative sea-level rise: wetland elevation trends and process controls. Ecological Studies 190:271–292.

California Coastal Sediment Management Workgroup (CCSMW), 2017. Eureka Littoral Cell-Coastal Regional Sediment Management Plan, p. 176.

California State Coastal Conservancy (CA-SCC) 2012. California Coastal Conservancy Coastal Lidar Project Bare Earth Lidar Digital Elevation Data.

Callaway, J. C., J. A. Nyman, and R. D. DeLaune. 1996. Sediment accretion in coastal wetlands: A review and a simulation model of processes. Current Topics in Wetland Biogeochemistry 2:2-23.

Costa, S. L. 1982. The physical oceanography of Humboldt Bay. Proceedings Humboldt Bay Symposium. C. Toole and C. Diebel (ed.), HSU Center for Community Development, Arcata, CA.

Curtis, J, A., Flint, L.E., Stern, M.A., Lewis, J., and Klein, R.D. in review. Fine-sediment supply under current and future climates for Humboldt Bay, CA, submitted to Estuaries and Coasts

D'Alpaos A., Mudd, S.M., and L. Carniello, 2011. Dynamic response of marshes to perturbations in suspended sediment concentrations and rates of relative sea level rise: Earth Surface, v. 116, no. F4,

Dettinger, M.D., Ralph, F.M., Das, T., Neiman, P.J. and Cayan, D.R., 2011. Atmospheric rivers, floods and the water resources of California. *Water*, *3*(2), pp.445-478.

Fagherazzi, S., Kirwan, M.L., Mudd, S.M., Guntenspergen, G.R., Temmerman, S., D'Alpaos, A., van de Koppel, J., Rybczyk, J.M., Reyes, E., Craft, C. and Clough, J., 2012. Numerical models of salt marsh evolution: Ecological, geomorphic, and climatic factors. *Reviews of Geophysics*, *50*(1).

Farnsworth, K.L., Warrick, J.A., 2007, Sources, Dispersal, and Fate of Fine Sediment Supplied to Coastal California: U.S. Geological Survey Scientific Investigations Report 2007–5254, 77 p.

Ganju, N. K., N. J. Nidzieko, and M. L. Kirwan, 2013. Inferring tidal wetland stability from channel sediment fluxes: Observations and a conceptual model, J. Geophys. Res. Earth Surf., 118, 2045.

Ganju, N. K., M. L. Kirwan, P. J. Dickhudt, G. R. Guntenspergen, D. R. Cahoon, and K. D. Kroeger, 2015. Sediment transport-based metrics of wetland stability, Geophys. Res. Lett., 42, 7992–8000.

Humboldt Bay Harbor Recreation and Conservation District (HBHRCD), 2015. Beneficial Reuse of Dredged Materials for Tidal Marsh Restoration and Sea Level Rise Adaptation in Humboldt Bay, California, Eureka, CA.

Kelsey, H.M., 1980. A sediment budget and an analysis of geomorphic process in the Van Duzen River basin, north coastal California, 1941–1975: summary. Geological Society of America Bulletin, Part I 91, 190–195.

Klein, R.D., Lewis, J. and Buffleben, M.S., 2012. Logging and turbidity in the coastal watersheds of northern California. *Geomorphology*, *139*, pp.136-144.

Kirwan, M. L., G. R. Guntenspergen, A. D'Alpaos, J. T. Morris, S. M. Mudd, and S. Temmerman (2010), Limits on the adaptability of coastal marshes to rising sea level, Geophys. Res. Lett., 37, L23401, doi:10.1029/2010GL045489.

Laird, A., B. Powell, J. Robinson and K. Shubert. 2007. Historical atlas of Humboldt Bay and the Eel River delta. Humboldt Bay Harbor, Recreation and Conservation District, Eureka, CA. Electronic document: DVD.

Laird, A. 2013. Humboldt Bay Shoreline Inventory, Mapping and Sea Level Rise Vulnerability Assessment. Prepared for State Coastal Conservancy. Trinity Associates, Arcata, CA.

Leithold, E.L., Perkey, D.W., Blair, N.E., Creamer, T.N., 2005. Sedimentation and carbon burial on the northern California continental shelf: the signatures of land-use change. Continental Shelf Research 25 (3), 349–371.

Lynch, J. C., P. Hensel, and D. R. Cahoon. 2015. The surface elevation table and marker horizon technique: A protocol for monitoring wetland elevation dynamics. Natural Resource Report NPS/NCBN/NRR—2015/1078. National Park Service, Fort Collins, Colorado.

Milliman, J.D., Farnsworth, K.L., 2011. River Discharge to the Coastal Ocean – A Global Synthesis. Cambridge University Press, Cambridge, UK, 384pp.

Montillet JP, Melbourne TI, Szeliga, WM (2018) GPS vertical land motion corrections to sea-level rise estimates in the Pacific Northwest. Journal of Geophysical Research: Oceans 123:1196−1212.

Morehead, M.D. and Syvitski, J.P. (1999) River-plume sedimentation modeling for sequence stratigraphy: application to the Eel margin, northern California. *Mar. Geol.*, **154**, 29–41.

Morris, J.T., P.V. Sundareshwar, C.T. Nietch, B. Kjerfve, and D.R. Cahoon. 2002. Responses of coastal wetlands to rising sea level. Ecology 83:2869-2877.

Orr, M., Crooks, S. and Williams, P.B., 2003. Will restored tidal marshes be sustainable? *San Francisco Estuary and Watershed Science*, *1*(1).

Patrick, W., & DeLaune, R., 1990. Subsidence, Accretion, and Sea Level Rise in South San Francisco Bay Marshes. *Limnology and Oceanography, 35*(6), 1389-1395.

Pethick JS. 1996. The geomorphology of mudflats. In: Nordstrom KF, Roman CT, editors. Estuarine shores: evolution, environment and human health. Chichester, UK: John Wiley. p 185-211.

Pethick JS, Crooks S. 2000. Development of a coastal vulnerability index: a geomorphological perspective. Environmental Conservation 27:359-367.

Pickart, A. 2001.The distribution of *Spartina densiflora* and two rare salt marsh plants in Humboldt Bay 1998-1999. Unpublished report, U.S. Fish and Wildlife Service, Humboldt Bay National Wildlife Refuge, Arcata, California.

Pickart, A. 2012. *Spartina densiflora* invasion ecology and the restoration of native salt marshes, Humboldt Bay, California. unpublished report, U.S. Fish & Wildlife Service, Arcata, California

Pickart, A, 2013, A Comparison of Mechanical Treatments for the Control of *Spartina densiflora* at Jacoby Creek Unit, Humboldt Bay National Wildlife Refuge**,** unpublished report, U.S. Fish & Wildlife Service, Arcata, California.

Rasmussen, P.P., Gray, J.R., Glysson, G.D., and Ziegler, A.C., 2009, Guidelines and procedures for computing time-series suspended-sediment concentrations and loads from in-stream turbidity-sensor and streamflow data: U.S. Geological Survey Techniques and Methods, book 3, chap. C4, 52 p.

Russell, N.; Griggs, G., 2012. Adapting to Sea Level Rise: A Guide for California's Coastal Communities. Prepared for the California Energy Commission Public Interest Environmental Research Program, University of California Santa Cruz. 50 pp.

Schlosser, S., and A. Eicher. 2012. The Humboldt Bay and Eel River Estuary Benthic Habitat Project. California Sea Grant Publication T-075, 246 p.

Stralberg, D., Brennan, M., Callaway, J.C., Wood, J., Schile, L., Jongsomjit, D., Kelly, M., Parker, V.T., and Crooks, S. (2011). Evaluating tidal marsh sustainability in the face of sea-level rise: a hybrid modeling approach applied to San Francisco Bay. PLOS ONE, 6.11: e27388.

Takekawa, J.Y., Thorne, K.M., Buffington, K.J., Freeman, C.M., Powelson, K.W., and Block G. 2013. Assessing marsh response from sea-level rise applying local site conditions: Humboldt Bay National Wildlife Refuge. Unpubl. Data Summary Report. USGS Western Ecological Research Center, Vallejo, CA. 44pp + Appendices.

Thom, R. M. 1992. Accretion rates of low intertidal salt marshes in the Pacific Northwest. Wetlands

Thorne, K.M., MacDonald, G.M., Ambrose, R.F., Buffington, K.J., Freeman, C.M., Janousek, C.N., Brown, L.N., Holmquist, J.R., Gutenspergen, G.R., Powelson, K.W., Barnard, P.L., and Takekawa, J.Y., 2016, Effects of climate change on tidal marshes along a latitudinal gradient in California: U.S. Geological Survey Open-File Report 2016-1125, 75 p.

Thorne, K.M., G. MacDonald, G. Guntenspergen, R. Ambrose, K. Buffington, B. Dugger, C. Freeman, C. Janousek, L. Brown, J. Rosencranz, J. Holmquist, J. Smol, K. Hargan, and J. Takekawa. 2018. U.S. Pacific coastal wetland resilience and vulnerability to sea-level rise: Science Advances, v. 4, no. 2.

van Proosdij, D., Lundholmb, J., Neatt, N., Bowron, T., Graham, J., 2010. Ecological reengineering of a freshwater impoundment for salt marsh restoration in a hypertidal system. Ecological Engineering 36, 1314e1332.

Waananen, A.O., Harris, D.D. and Williams, R.C., 1971. *Floods of December 1964 and January 1965 in the Far Western States; Part 1 Description* (No. 1866-A). US Govt. Print. Office. 276p.

Wagner, R.J., Boulger, R.W., Jr., Oblinger, C.J., and Smith, B.A., 2006, Guidelines and standard procedures for continuous water-quality monitors—Station operation, record computation, and data reporting: U.S. Geological Survey Techniques and Methods 1–D3, 51 p.

Warrick, J.A., Madej, M.A., Goñi, M.A., Wheatcroft, R.A., 2013. Trends in the suspended sediment yields of coastal rivers of northern California, 1955–2010. Journal of Hydrology 489, 108–123.

Warrick, J.A., 2014. Eel River margin source-to-sink sediment budgets: Revisited. *Marine Geology*, *351*, pp.25-37.

Webb, E.L., Friess, D.A., Krauss, K.W., Cahoon, D.R., Guntenspergen, G.R., and Phelps, J., 2013, A global standard for monitoring coastal wetland vulnerability to accelerated sea-level rise: Nature Climate Change, v. 3, p. 458–465.

Wheatcroft, R.A., Sommerfield, C.K., Drake, D.E., Borgeld, J.C., Nittrouer, C.A., 1997. Rapid and widespread dispersal of flood sediment on the northern California margin. Geology 25, 163–166.

Wheatcroft, R.A., Borgeld, J.C., 2000. Oceanic flood deposits on the northern California shelf: large-scale distribution and small-scale physical properties. Continental Shelf Research 20, 2163–2190.