Estuarine, Coastal and Shelf Science 243 (2020) 106910

Contents lists available at ScienceDirect

Estuarine, Coastal and Shelf Science

journal homepage: http://www.elsevier.com/locate/ecss

Large-scale erosion driven by intertidal eelgrass loss in an estuarine environment

Ryan K. Walter[a,*], Jennifer K. O'Leary[b,c], Sean Vitousek[d,e], Mohsen Taherkhani[e], Carolyn Geraghty[f], Ann Kitajima[f]

[a] Physics Department, California Polytechnic State University, San Luis Obispo, CA, USA
[b] California Sea Grant, San Luis Obispo, CA, USA
[c] Wildlife Conservation Society, Mombasa, Kenya
[d] United States Geological Survey, Pacific Coastal and Marine Science Center, Santa Cruz, CA, USA
[e] Department of Civil and Materials Engineering, University of Illinois, Chicago, IL, USA
[f] Morro Bay National Estuary Program, Morro Bay, CA, USA

ARTICLE INFO

Keywords:
Seagrass
Estuarine geomorphology
Estuarine hydrodynamics
Ecosystem collapse
Ecological phase shift
Habitat loss

ABSTRACT

Seagrasses influence local hydrodynamics by inducing drag on the flow and dampening near-bed velocities and wave energy. When seagrasses are lost, near-bed currents and wave energy can increase, which enhances bottom shear stresses, destabilizes sediment, and promotes suspension and erosion. Though seagrasses are being lost rapidly globally, the magnitude of change in sediment stabilization following ecosystem-wide eelgrass loss has rarely been measured. In this study, we explored the geomorphological changes associated with an unprecedented estuary-wide collapse of a seagrass (eelgrass, *Zostera marina*) in Morro Bay, CA, USA. Morro Bay has historically suffered from accelerated sedimentation and accretion. However, following massive eelgrass loss since 2010, over 90% of locations that previously had eelgrass experienced erosion. Elevation losses (erosion) reached 0.50 m in some places (mean loss of 0.10 m) with as much as a 50% decrease (median decrease of 13.6%) in elevation (i.e., increase in depth) compared to pre-decline levels. In comparison, the mouth of the estuary, where eelgrass was largely retained, had only 27.7% of the locations with prior eelgrass experiencing erosion and underwent a mean elevation increase (accretion) of 0.32 m. Thus, the loss of eelgrass appears to have altered dynamics at the seabed and transitioned large regions of the estuary from an environment that promotes deposition and accretion to one that promotes suspension and erosion. Large-scale erosion following seagrass loss may be predictive of future shoreline and coastal habitat changes and is likely to be exacerbated by increased storm surge and sea level rise expected with climate change.

1. Introduction

Terrestrial and marine environments are under increasing threat from climate change, pollution, and other anthropogenic influences (Halpern et al., 2008; Dirzo et al., 2014). Marine habitat loss is predicted to rapidly intensify over the next century, likely leading to shifts in ecosystem state and loss of marine fauna (McCauley et al., 2015). In shallow coastal and estuarine marine habitats, seagrass meadows are declining at an alarming rate, with loss rates comparable to those reported for tropical rainforests, mangroves, and coral reefs (Orth et al., 2006; Waycott et al., 2009). Losses of seagrasses can have a substantial impact on shallow marine ecosystems since seagrasses provide many ecosystem services, including fish nursery habitats (Beck et al., 2001), forage for migratory birds (Shaughnessy et al., 2012), nutrient cycling (McGlathery et al., 2007), carbon storage (Duarte et al., 2005), and sediment stabilization (Hansen and Reidenbach, 2012).

Seagrasses are considered ecosystem engineers because they beneficially modify the local biological, chemical, and physical environment through self-sustaining positive feedbacks (Maxwell et al., 2017). One such example is the well-documented seagrass-sediment-light feedback, which has been described as one of the most important feedbacks in seagrass ecosystems (de Boer, 2007; Adams et al., 2016; Moksnes et al., 2018). Seagrasses influence local hydrodynamics by inducing drag on the flow and dampening near-bed velocities and wave energy (Fonseca

* Corresponding author.
E-mail address: rkwalter@calpoly.edu (R.K. Walter).

https://doi.org/10.1016/j.ecss.2020.106910
Received 8 April 2020; Received in revised form 22 June 2020; Accepted 23 June 2020
Available online 26 July 2020
0272-7714/© 2020 Elsevier Ltd. All rights reserved.

et al., 1982; Koch et al., 2006; Lacy and Wyllie-Echeverria, 2011; Gacia and Duarte, 2001; Hansen and Reidenbach, 2012, 2013; Reidenbach and Thomas, 2018; Paquier et al., 2019). This in turn, affects the bottom boundary layer and reduces shear stresses near the bed that are responsible for sediment suspension, thereby promoting a depositional environment and sediment retention (Ward et al., 1984; Hansen and Reidenbach, 2012, 2013; Reidenbach and Thomas, 2018). Sediment stabilization by the seagrasses decreases turbidity and hence improves light penetration to the benthos, which in turn promotes seagrass photosynthesis and growth (Lawson et al., 2007). This positive feedback between seagrass, sediment, and light has been documented in individual seagrass beds and meadows, with seagrass meadow and environmental characteristics influencing the strength of the feedback (cf. van der Heide et al., 2011; Table 2 in Adams et al. 2016; Aoki et al., 2020).

Conversely, when seagrasses are lost, near-bed currents and wave energy increase, which enhances bottom shear stresses, thereby destabilizing sediment and causing erosion. This process increases turbidity in the water column and decreases light availability, conditions unfavorable for seagrass growth and survival. Thus, in addition to the initial stressor(s) contributing to seagrass loss (e.g., eutrophication, high temperature, sediment quality and quantity, etc.; see overview in Orth et al., 2006), the concurrent loss in ecosystem engineering benefits can lead to further collapse. In some cases, this can lead to an unvegetated alternative state where seagrass recovery is inhibited and erosion is favored over deposition. The resulting geomorphological changes can range in scale from the size of a seagrass bed/meadow to an entire system, though the latter is not as widely documented (e.g., Wilson, 1949; Christiansen et al., 1981; van Der Heide et al., 2007; Maxwell et al., 2017; Moksnes et al., 2018).

Comprehensive before and after data are not always available in ecosystem collapses, as major ecosystem changes are often unforeseen. In Morro Bay, a short, shallow estuary along the California coast (USA), there was an estuary-wide seagrass (eelgrass, *Zostera marina*) loss that occurred over a several year period. We used bathymetry and topography data collected before and after the loss of seagrass to explore geomorphological changes and show that the altered system is consistent with the hypothesis that the loss of seagrass altered the dynamics of the seafloor to favor an erosional environment.

2. Study site and methods

2.1. Morro Bay, California, USA

Morro Bay is a shallow estuary located along the Central California Coast (Fig. 1a). It is home to a major fishing port for local fisheries and two shellfish aquaculture facilities, and it features a diverse population of fish, invertebrates, and birds (both native and migratory). Morro Bay is a seasonally low-inflow estuary (LIE) with a Mediterranean climate typified by an extended dry season [~April to October, with a mean (standard deviation) monthly precipitation of 1.17 cm (2.27 cm) from 1988 to 2019 during these months; see Section 2.4 for data details] with little to no precipitation and freshwater inputs, and a shorter wet season [~November to March, with a mean monthly precipitation of 8.53 cm (8.96 cm) from 1988 to 2019 during these months; see Section 2.4 for data details] with episodic rainfall and freshwater inputs. This tidally forced estuary is characterized by a main channel that runs the length of the bay (~6.5 km) and becomes progressively shallower going from the mouth to the head.

The main subtidal channel is flanked by intertidal flats, which historically supported expanses of eelgrass (*Zostera marina*), a temperate seagrass. However, from 2007 to 2017, intertidal eelgrass in Morro Bay declined from 139.2 ha to 5.4 ha, with the majority of losses occurring from 2010 to 2013 in the intertidal flats in the mid-to back-bay regions and the remaining eelgrass beds located near the mouth (Fig. 1b; data described in Section 2.2). While the cause of the decline is not clear, recent research following the decline showed that there are strong gradients in environmental conditions throughout the bay (Walter et al., 2018a). In particular, the mid to back portions of the estuary were found to have significantly higher turbidities and longer flushing times compared to the mouth, although it is not clear to what extent conditions were altered following the eelgrass decline (Walter et al., 2018a).

Morro Bay has been heavily modified over the last century, including the closing of one of its natural entrances in the early 1900s, construction of several breakwaters and a dike in the 1940s, and dredging of the mouth that has increased in frequency over the last several decades (CCWQCB, 2002). Historically, Morro Bay has suffered from accelerated sedimentation, and in 1998 the Central Coast Water Quality Control Board (CCWQCB) identified the estuary as being impaired by accelerated sedimentation/siltation and listed the water body under the Clean Water Act Section 303(d) (CCWQCB, 2002). Over the last century (1884–1987), the tidal prism (mean volume of water between mean

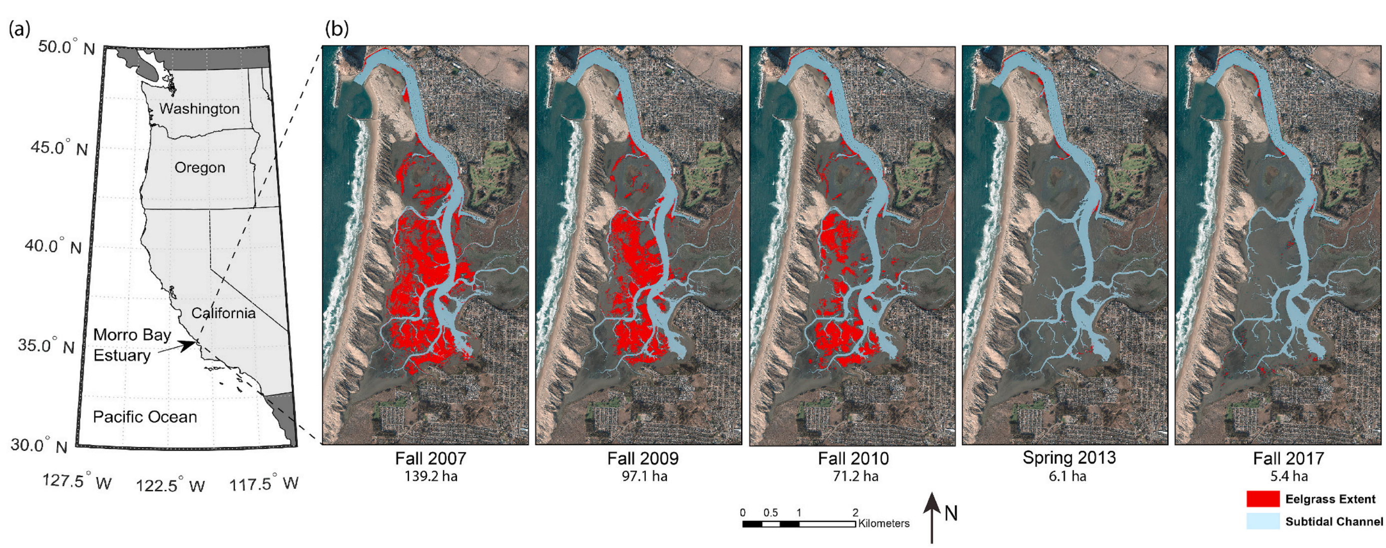

Fig. 1. (a) West Coast of the United States with the location of the Morro Bay, CA, estuary. (b) Spatial distribution of eelgrass coverage (red) over time, highlighting the recent large-scale collapse of the major biogenic habitat in Morro Bay from Fall 2007 to Fall 2017 with the eelgrass area shown under the date on each image. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

high and low tides) in Morro Bay has decreased by approximately 25% due to more than half a meter of accretion of the intertidal mudflats (Haltiner et al., 1991). Intertidal accretion is estimated to have resulted in the loss of over 140 ha of potential eelgrass habitat based on observed upper and lower depth limits of eelgrass (Chestnut, 1999). Moreover, out of seven sites assessed in California and Mexico, the salt marsh in Morro Bay had the highest observed accretion rates of any site (Thorne et al., 2016). Finally, a model with a spatially and temporally uniform rate of sedimentation projected that over the next century, Morro Bay will exhibit sediment-induced elevation changes that will lead to significant losses in eelgrass habitat (Shaughnessy et al., 2012).

2.2. Historical eelgrass data

Historical eelgrass coverage has been monitored intermittently since 1960 (MBNEP, 2017). Until 2002, coverage was estimated using a combination of field surveys and aerial photos. Starting in 2002, eelgrass distributions were mapped periodically in the late fall (with the exception being the spring of 2013) using aerial multispectral imagery taken at extreme low tides, since most of the eelgrass is intertidal (see MBNEP, 2017). Guided classification methods were used for preliminary classification of the eelgrass, with follow up field surveys to verify classifications (MBNEP, 2017).

2.3. Bathymetry and topography data

To assess recent geomorphological changes related to the eelgrass collapse, bathymetry and topography data for the Morro Bay region from before and after the eelgrass loss were obtained from the NOAA Office of Coastal Management (https://coast.noaa.gov/dataviewer/#/lidar/search/-13457732.510972984,4205259.548137256,-13449400.874889899,4216419.354266891). The first elevation dataset (hereafter referred to as the 2010 survey for brevity; https://inport.nmfs.noaa.gov/inport/item/49649) was collected when intertidal eelgrass coverage throughout the bay was at its recent peak. This dataset is a 1 m resolution Digital Elevation Model (DEM) that combines the following datasets (collected between 2009 and 2011): topographic Lidar from the California Coastal Conservancy Lidar Project, bathymetric Lidar from the California Coastal Mapping Project and Army Corps of Engineers Joint Airborne Lidar Bathymetry Center of Expertise, and multibeam acoustic data from the California Seafloor Mapping Program. We commissioned the collection of the second dataset (hereafter referred to as the 2019 survey for brevity; https://inport.nmfs.noaa.gov/inport/item/57916) to be able to assess geomorphological changes following the eelgrass decline. These data were collected in the late spring and early summer of 2019. This dataset is also a 1 m resolution DEM that combines Lidar (both traditional near-infrared with green wavelength for bathymetric Lidar) and multibeam acoustic data. For the intertidal regions of the bay, the topographic Lidar had vertical accuracies [i.e., root mean square errors (RMSE) when compared to GPS survey grade points] of 4.8 cm and 1.8 cm for the 2010 and 2019 surveys, respectively. For subtidal regions, the 2019 survey had a RMSE of 5.2 cm, while the 2010 survey had a RMSE of 15 cm for the bathymetric Lidar and the NOAA metadata indicates a varied vertical accuracy for the multibeam acoustic survey since these data came from multiple datasets (expected to be on the order of tens of centimeters based on NOAA standards). Both datasets had the same horizontal (NAD83) and vertical datum, where an elevation of zero represents mean sea level.

Absolute changes in elevation were quantified by the difference between the post-decline elevations (z_{2019}) and pre-decline elevations (z_{2010}), where a positive or negative change represents accretion or erosion, respectively. Relative percent changes in elevation were quantified as,

$$\%\ change = \frac{z_{2019} - z_{2010}}{|z_{2010}|} \times 100, \tag{1}$$

where the absolute value in the denominator accounts for the negative elevations in the estuary (i.e., below mean sea level). In Equation (1), a positive percent change in elevation represents a gain of elevation relative to the mean sea level through accretion, or a decrease in the local depth. A negative percent change in elevation represents a loss of elevation relative to the mean sea level through erosion, or an increase in the local depth.

2.4. Other environmental data

To provide environmental context for the eelgrass loss and resulting geomorphological changes observed, we synthesized long-term environmental datasets (e.g., precipitation, dredging records, and temperature). Local precipitation data were obtained from the California Irrigation Management Information System (CIMIS) station 52 (35.305442°N, 120.66178°W; https://cimis.water.ca.gov/Stations.aspx), which is the longest (1986 – present) and most complete record in the vicinity of Morro Bay. Historical dredging volume records from the mouth of the bay were obtained from the Army Corps of Engineers since 1986. Finally, ocean temperature measurements were obtained going back to 2007 from a long-term monitoring site at the mouth of the estuary (https://www.cencoos.org/data/shore/morro).

3. Results

3.1. Recent eelgrass collapse

Fig. 1b shows the spatial distribution of eelgrass coverage throughout the bay when available from 2007 to 2017. Eelgrass area declined from 139.2 ha to 97.1 ha–71.2 ha in just a three year period (Fall 2007, Fall 2009, Fall 2010, respectively). There were some smaller areas that showed considerable year-to-year variability between Fall 2007 and Fall 2010 with some areas showing small amounts of growth between 2007 and 2010 (Fig. 1b; mid-bay, west side of the main channel), although the general large-scale trend demonstrated a sizable loss of eelgrass. After 2010, the decline in eelgrass accelerated, with intertidal acreage plummeting to 6.1 ha by the Spring of 2013, with similar levels (5.4 ha) observed in the Fall of 2017. Most of the intertidal eelgrass in the mid-to back-bay was no longer present in 2013 nor in 2017, with the remaining eelgrass located near the mouth and along the edges of the main channel.

3.2. Historical variability and environmental context

The historical acreage of eelgrass in Morro Bay has displayed considerable interannual variability (Fig. 2a). However, the most recent decline to 5.4 ha is unprecedented. In a previous decline from 1994 to 1998, eelgrass acreage in the bay went from 176.0 ha to 39.7 ha. This decline coincided with a large fire in the surrounding watershed in August 1994 that was followed by several of the largest annual precipitation years over the last four decades (e.g., 1995, 1996) that deposited significant amounts of fine sediment into the back portions of the estuary (Fig. 2b). This period also had the largest dredging event on record in 1995 (Fig. 2c). However, the 1994 to 1998 decline was followed by a rebound of eelgrass close to pre-decline levels in the following years.

In the recent decline, the most substantial losses occurred from 2010 to 2013, and 2010 coincided with a relatively wet year with large annual precipitation. There was also a dredging event in late 2009 that extended past the mouth (red bar in Fig. 2c). However, after 2010, annual precipitation (and sediment inputs) and dredging were similar to pre-decline years. Seasonal temperature ranges and summer temperature maximums in the bay did not change substantially between 2007

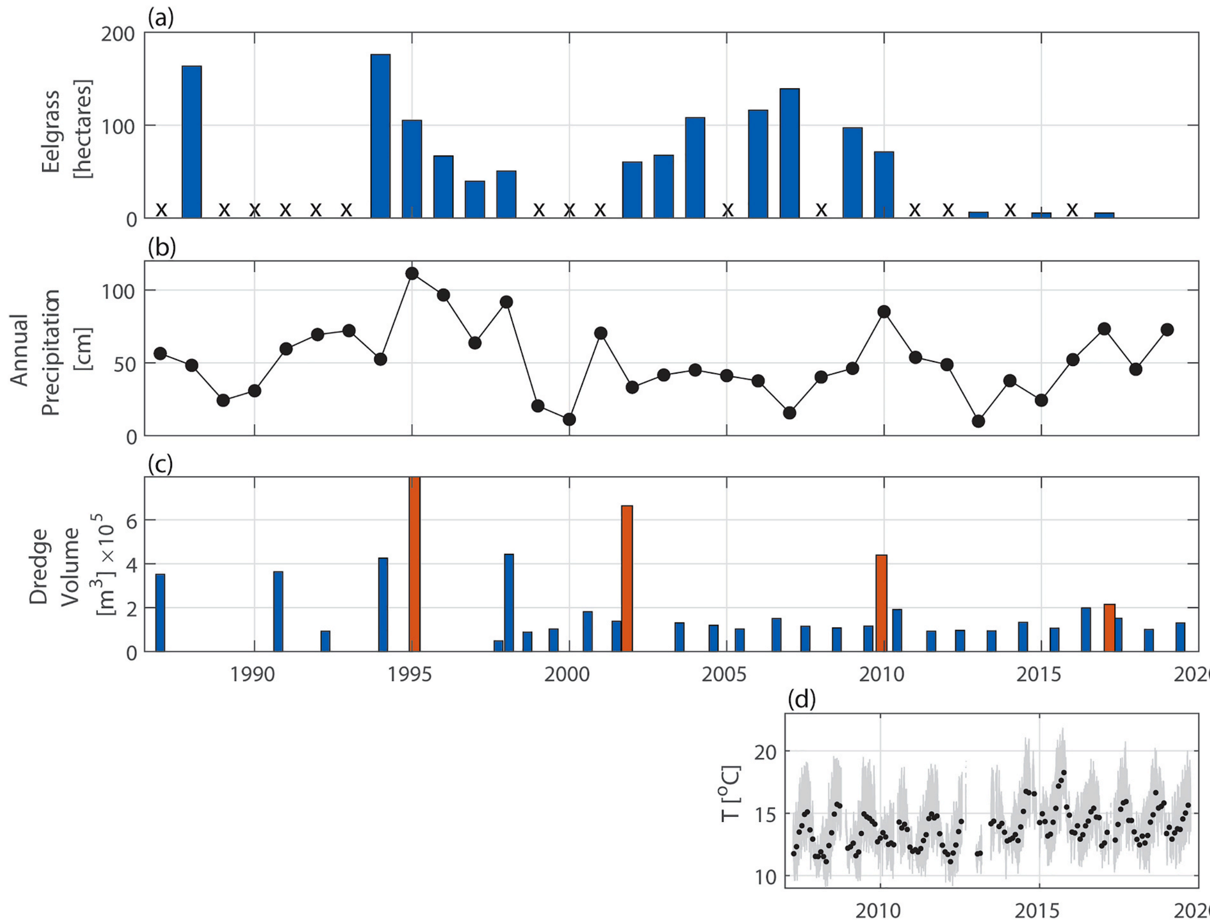

Fig. 2. Historical data on (a) eelgrass acreage, (b) annual precipitation from the surrounding watershed, (c) dredging volume near the mouth of the estuary, and (d) water temperature at the mouth of the estuary. The small "x" in panel (a) denote years where no eelgrass surveys were completed. In panel (c) dredging data are shown according to their start date. The red bins indicate dredging events that extended further into the bay relative to the blue bin dredging events (see Fig. 3a, white arrows, for endpoint). The gray lines in panel (d) denote the raw data and the black dots represent monthly averages. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

and 2013 and followed the well-established seasonal cycles for this region (Fig. 2d; cf. Walter et al., 2018b; Barth et al., 2020). Temperatures did increase from 2014 to 2016 with the Northeast Pacific Marine Heatwave ("warm blob" followed by an El Niño; e.g., Gentemann et al., 2016), but this occurred after the major period of eelgrass collapse. The other dominant low-frequency climate modes, in addition to El Niño Southern Oscillation (ENSO), that have been shown to drive substantial interannual variability in the Pacific Ocean, the North Pacific Gyre Oscillation (NPGO) and Pacific Decadal Oscillation (PDO), underwent long-term phase shifts in late 2013 and early 2014, respectively (e.g., Fig. 3 in Barth et al., 2020). However, these phase shifts also occurred after the initial decline and major period of eelgrass collapse.

3.3. Bathymetry and topography changes and eelgrass

In comparing the elevation changes between 2010 (pre-decline survey) and 2019 (post-decline survey), the largest values of accretion and erosion were near the mouth of the bay, which is exposed to more energetic forcing (waves and tidal currents) and subject to annual dredging (Fig. 3b). There were also regions of smaller amounts of accretion and erosion throughout the main channel. However, the main change in the estuary was the large-scale and spatially consistent erosion that occurred over the majority of the intertidal flats in areas that previously supported large expanses of eelgrass meadows (Figs. 3b and 1b). Using the historical distribution of eelgrass coverage before the decline (i.e., composite of polygons from 2007, 2009, and 2010 surveys), we calculated the corresponding changes in elevation for areas formerly occupied by eelgrass. At these locations ($n = 1.506 \times 10^6$ grid points), 90.72% of the points underwent erosion, with a mean elevation change of -0.10 m (i.e., negative implies erosion), a standard deviation of 0.13 m, and some losses reaching 0.50 m (see Fig. 3d for distribution). In some regions, erosional losses decreased the elevation (i.e., increased the depth) of the intertidal flats that previously had eelgrass by up to 50% relative to pre-decline levels with a median decrease in elevation of 13.6% (Fig. 3c; see Fig. 3e for distribution).

While the mid-to back-bay intertidal portions of the estuary experienced erosion, the mouth of the estuary (i.e., northing $> 3.91585 \times 10^6$; Fig. 3a), where eelgrass was largely retained, eroded at only 27.68% of the sites with eelgrass. At the mouth, there was a mean elevation change of $+0.32$ m (i.e., accretion) with a standard deviation of 0.66 m (i.e., much larger variability).

4. Discussion and conclusions

The large-scale loss of eelgrass in Morro Bay provides a unique opportunity to assess the role of seagrasses on sediment retention in an entire estuarine system and test key hypotheses related to the seagrass-sediment-light feedback. Comparing elevations throughout the bay from bathymetry and topography surveys before and after the estuary-wide loss showed that in places that previously had eelgrass, over 90% of the points underwent a loss of elevation (i.e., erosion), with elevation losses reaching 0.50 m and nearly a 50% decrease in elevation (i.e.,

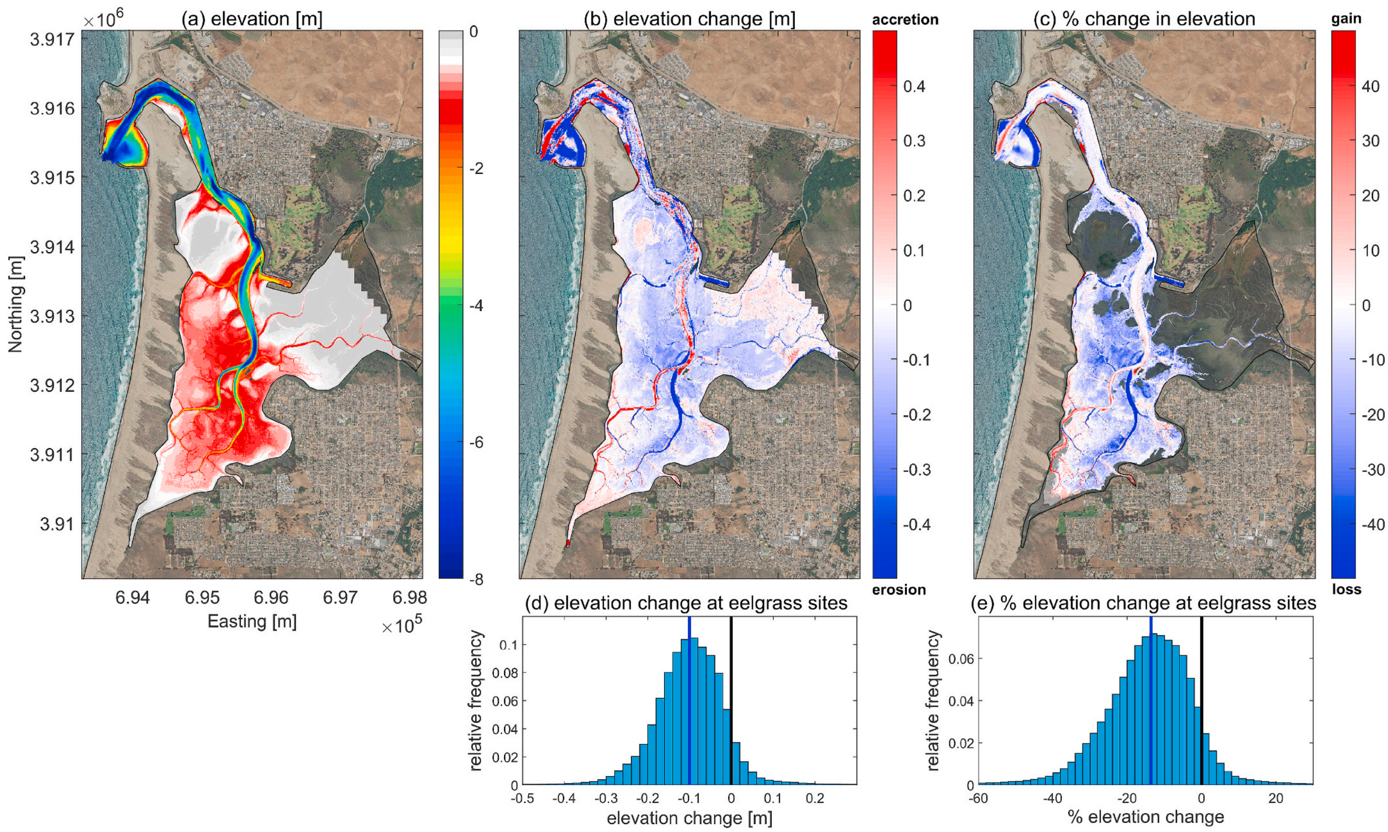

Fig. 3. (a) Elevation relative to mean sea level from the most recent (2019) survey. (b) Change in elevation between 2010 and 2019 ($z_{2019} - z_{2010}$), where red areas denote accretion and blue areas represent erosion. (c) Percent change in elevation between 2010 and 2019 (Equation (1)), where red areas denote gains and blue areas represent losses. (d) Relative frequency distribution of the change in elevation between 2010 and 2019 at sites that previously had eelgrass (composite of points from 2007, 2009, and 2010 surveys), with the mean change shown as a solid blue vertical line. (e) Relative frequency distribution of the percent change in elevation between 2010 and 2019 (Equation (1)) at sites that previously supported eelgrass (composite of points from 2007, 2009, and 2010 surveys), with the median percent change shown as a solid blue vertical line. In panel (a), the white arrows point to the solid white lines in the main channel that represent the end point of dredging at the mouth for the blue (northernmost arrow) and red (southernmost arrow) bins in Fig. 2. The gray shading between −0.5 m and 0 m elevation in panel (a) highlights elevations shallower than the upper depth limit for eelgrass based on pre-decline surveys (i.e., 90th percentile of −0.48 m elevation for eelgrass elevation range from 2007, 2009, and 2010 surveys). These same locations are not shown in panel (c) for consistency. The solid black line in panel (a) denotes the cutoff point chosen for the mouth of the estuary in Section 3.3 (i.e., northing of 3.91585×10^6). In panels (d) and (e), the histograms are normalized to show the relative frequency in each bin such that the sum of the bins is equal to one, and the solid black vertical line is shown at zero change for reference. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

increase in depth) compared to pre-decline levels (see distributions in Fig. 3d and e, respectively). These losses are significant considering that prior to the eelgrass decline, multiple studies showed that Morro Bay historically suffered from accelerated sedimentation (i.e., accretion) throughout the estuary and adjacent salt marsh (see Section 2 for details; CCWQCB, 2002; Haltiner et al., 1991; Chestnut, 1999; Thorne et al., 2016). Thus, the loss of eelgrass appears to have altered the dynamics at the seabed in places that previously held vegetation and transitioned large regions of the estuary from an environment that promotes deposition and accretion to one that promotes suspension and erosion.

Following the loss of seagrasses, many systems transition to unvegetated alternative states, where the loss of seagrass ecosystem engineering benefits (e.g., seagrass-sediment-light feedback) can inhibit recovery (van Der Heide et al., 2007; Maxwell et al., 2017; Moksnes et al., 2018). In Morro Bay, a hydrodynamic study following the decline found that the back portions of the estuary, where the majority of eelgrass loss occurred, had elevated turbidities relative to the mouth (Walter et al., 2018a). This finding supports the idea that eelgrass loss in the mid to back portions of the estuary resulted in the transition to an environment that fosters sediment suspension and erosion, possibly resulting in elevated turbidities.

Preliminary data and analysis from drone-based surveys throughout the bay from December 2019 indicated a small rebound in eelgrass from 5.4 ha in December 2017 to 14.9 ha in December 2019 (Walter et al., unpublished data). While this increase to 14.9 ha is still only 10.7% of the pre-decline area (139.2 ha from 2007), this partial recovery is a positive sign and warrants further investigation. Projects are underway to look at the success of small-scale experimental outplants at various locations and depths, as well as the development of a high-resolution hydrodynamic numerical model to further understand sediment dynamics in the bay. It is possible, though untested, that erosion following seagrass loss has created new opportunities for seagrass recovery by increasing the depth and potential suitable habitat for eelgrass in certain locations. However, it is also feasible that some portions of the bay will not recover due to changes in suspended sediment and light conditions and may be trapped in a negative feedback loop where loss of eelgrass creates conditions that prevent regrowth.

Seagrass ecosystems offer a natural means of shoreline and sediment stabilization (cf. Ondiviela et al., 2014; Boudouresque et al., 2016; Paquier et al., 2019). This study demonstrates large-scale erosion following seagrass loss, which may be predictive of future shoreline change. Globally, seagrass systems are declining at an alarming rate and are among the most threatened ecosystems on the planet (Waycott et al., 2009). Seagrass loss is likely to significantly increase shoreline and estuarine erosion. Further, increased sea level rise and changes to storm surges expected with climate change will likely increase the frequency of coastal flooding and the erosion of shorelines (Vitousek et al., 2017a, 2017b; Taherkhani et al., 2020), and losses of seagrasses could exacerbate shoreline degradation. Thus, protection and restoration of seagrass is of paramount importance. There is a need for global, regional, and local improvements in spatial assessments of seagrass habitat, along with seagrass health and risk assessments to allow early warnings of seagrass decline and rapid management measures (Unsworth et al., 2019). Proactively working to prevent seagrass loss will help prevent regime shifts to unvegetated ecosystem states and contribute to climate change mitigation through sediment stabilization, shoreline protection, and carbon sequestration.

CRediT authorship contribution statement

Ryan K. Walter: Conceptualization, Methodology, Formal analysis, Investigation, Writing - original draft, Writing - review & editing, Visualization, Supervision, Project administration, Funding acquisition. **Jennifer K. O'Leary:** Conceptualization, Writing - original draft, Writing - review & editing, Project administration, Funding acquisition. **Sean Vitousek:** Conceptualization, Methodology, Investigation, Writing - review & editing, Visualization. **Mohsen Taherkhani:** Methodology, Writing - review & editing. **Carolyn Geraghty:** Investigation, Writing - review & editing. **Ann Kitajima:** Investigation, Writing - review & editing, Supervision, Project administration, Funding acquisition.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgments

This publication was prepared under NOAA Grant #NA18OAR4170073, California Sea Grant College Program Project #R/HCE-07, through NOAA's National Sea Grant College Program, U.S. Department of Commerce. The statements, findings, conclusions and recommendations are those of the authors and do not necessarily reflect the views of California Sea Grant, NOAA, or the U.S. Dept. of Commerce. We also acknowledge support from the NOAA IOOS program through CeNCOOS for long-term data collected at the Bay Mouth site. Bathymetry and topography data from the 2019 survey were supported by the Environmental Protection Agency, US Fish and Wildlife, and the NOAA Office of Coastal Management. Dredging data were obtained from Joe Ryan at the Army Corps of Engineers. We also acknowledged funding support from the U.S. Geological Survey's Coastal and Marine Hazards and Resources Program. We thank Renee Takesue for providing a USGS internal review of the manuscript, as well as the comments and suggestions of two anonymous reviewers that greatly improved the manuscript.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecss.2020.106910.

References

Adams, M.P., Hovey, R.K., Hipsey, M.R., Bruce, L.C., Ghisalberti, M., Lowe, R.J., Gruber, R.K., Ruiz-Montoya, L., Maxwell, P.S., Callaghan, D.P., Kendrick, G.A., O'Brien, K.R., 2016. Feedback between sediment and light for seagrass: Where is it important? Limnol. Oceanogr. 61, 1937–1955. https://doi.org/10.1002/lno.10319.

Aoki, L.R., McGlathery, K.J., Wiberg, P.L., Al-Haj, A., 2020. Depth affects seagrass restoration success and resilience to marine heat wave disturbance. Estuar. Coast 43, 316–328. https://doi.org/10.1007/s12237-019-00685-0.

Barth, A., Walter, R.K., Robbins, I., Pasulka, A., 2020. Seasonal and interannual variability of phytoplankton abundance and community composition on the Central Coast of California. Mar. Ecol. Prog. Ser. 637, 29–43. https://doi.org/10.3354/meps13245.

Beck, M.W., Heck, K.L., Able, K.W., Childers, D.L., Eggleston, D.B., Gillanders, B.M., Halpern, B., Hays, C.G., Hoshino, K., Minello, T.J., Orth, R.J., Sheridan, P.F., Weinstein, M.P., 2001. The identification, conservation, and management of estuarine and marine nurseries for fish and invertebrates. Bioscience 51, 633. https://doi.org/10.1641/0006-3568(2001)051[0633:ticamo]2.0.co;2.

Boudouresque, C.F., Pergent, G., Pergent-Martini, C., Ruitton, S., Thibaut, T., Verlaque, M., 2016. The necromass of the Posidonia oceanica seagrass meadow: fate, role, ecosystem services and vulnerability. Hydrobiologia 781, 25–42. https://doi.org/10.1007/s10750-015-2333-y.

Central Coast Water Quality Control Board, 2002. Morro bay total maximum daily load for sediment (including Chorro Creek, Los Osos Creek and the Morro bay estuary). Retrieved. https://www.waterboards.ca.gov/centralcoast/board_decisions/adopted_orders/2002/2002_0051_mb_sed_tmdl_final_proj_rpt.pdf. (Accessed July 2018).

Chestnut, J., 1999. Eelgrass habitat in Morro Bay. Report to the Morro Bay National Estuary Program.

Christiansen, C., Christoffersen, H., Dalsgaard, Jø, NØrnberg, P., 1981. Coastal and near-shore changes correlated with die-back in eel-grass (Zostera Marina, L.). Sediment. Geol. 28, 163–173. https://doi.org/10.1016/0037-0738(81)90063-4.

de Boer, W.F., 2007. Seagrass–sediment interactions, positive feedbacks and critical thresholds for occurrence: a review. Hydrobiologia 591, 5–24. https://doi.org/10.1007/s10750-007-0780-9.

Duarte, C.M., Middelburg, J.J., Caraco, N., 2005. Major role of marine vegetation on the oceanic carbon cycle. Biogeosciences 2, 1–8. https://doi.org/10.5194/bg-2-1-2005.

Dirzo, R., Young, H.S., Galetti, M., Ceballos, G., Isaac, N.J.B., Collen, B., 2014. Defaunation in the anthropocene. Science 345, 401–406. https://doi.org/10.1126/science.1251817.

Fonseca, M.S., Fisher, J.S., Zieman, J.C., Thayer, G.W., 1982. Influence of the seagrass, Zostera marina L., on current flow. Estuar. Coast Shelf Sci. 15, 351–364. https://doi.org/10.1016/0272-7714(82)90046-4.

Gacia, E., Duarte, C.M., 2001. Sediment retention by a Mediterranean posidonia oceanica meadow: the balance between deposition and resuspension. Estuar. Coast Shelf Sci. 52, 505–514. https://doi.org/10.1006/ecss.2000.0753.

Gentemann, C.L., Fewings, M.R., García-Reyes, M., 2016. Satellite sea-surface temperatures along the west coast of the United States during the 2014-2016 northeast Pacific marine heat wave. Geophys. Res. Lett. 312–319. https://doi.org/10.1002/2016GL071039.

Halpern, B.S., Walbridge, S., Selkoe, K.A., Kappel, C.V., Micheli, F., D'Agrosa, C., Bruno, J.F., Casey, K.S., Ebert, C., Fox, H.E., Fujita, R., Heinemann, D., Lenihan, H.S., Madin, E.M.P., Perry, M.T., Selig, E.R., Spalding, M., Steneck, R., Watson, R., 2008. A global map of human impact on marine ecosystems, 80- Science 319, 948. https://doi.org/10.1126/science.1149345. LP – 952.

Haltiner, J.P., ASCE, Thor, D., 1991. Sedimentation process in Morro bay, California. In: Proceedings: Coastal Sediments Conference, ASCE Waterway, Ports, Coastal, and Ocean Division. Seattle, WA, June 1991.

Hansen, J.C., Reidenbach, M.A., 2013. Seasonal growth and senescence of a Zostera marina seagrass meadow alters wave-dominated flow and sediment suspension within a coastal bay. Estuar. Coast 36, 1099–1114. https://doi.org/10.1007/s12237-013-9620-5.

Hansen, J.C., Reidenbach, M.A., 2012. Wave and tidally driven flows in eelgrass beds and their effect on sediment suspension. Mar. Ecol. Prog. Ser. 448, 271–287. https://doi.org/10.3354/meps09225.

Koch, E.W., Sanford, L.P., Chen, S.-N., Shafer, D.J., Smith, J.M., 2006. Waves in Seagrass Systems: Review and Technical Recommendations. United States Army Corps of Engineers. Technical Report, ERDC TR-06-15.

Lacy, J.R., Wyllie-Echeverria, S., 2011. The influence of current speed and vegetation density on flow structure in two macrotidal eelgrass canopies. Limnol. Oceanogr. Fluid. Environ. 1, 38–55. https://doi.org/10.1215/21573698-1152489.

Lawson, S.E., Wiberg, P.L., McGlathery, K.J., Fugate, D.C., 2007. Wind-driven sediment suspension controls light availability in a shallow coastal lagoon. Estuar. Coast 30, 102–112. https://doi.org/10.1007/BF02782971.

Maxwell, P.S., Eklöf, J.S., van Katwijk, M.M., O'Brien, K.R., de la Torre-Castro, M., Boström, C., Bouma, T.J., Krause-Jensen, D., Unsworth, R.K.F., van Tussenbroek, B. I., van der Heide, T., 2017. The fundamental role of ecological feedback mechanisms for the adaptive management of seagrass ecosystems – a review. Biol. Rev. 92, 1521–1538. https://doi.org/10.1111/brv.12294.

McCauley, D.J., Pinsky, M.L., Palumbi, S.R., Estes, J.A., Joyce, F.H., Warner, R.R., 2015. Marine defaunation: animal loss in the global ocean. Science 347. https://doi.org/10.1126/science.1255641.

McGlathery, K.J., Sundbäck, K., Anderson, I.C., 2007. Eutrophication in shallow bays and lagoons: the role of plants in the coastal filter. Mar. Ecol. Prog. Ser. 348, 1–18. https://doi.org/10.3354/meps07132.

Moksnes, P.O., Eriander, L., Infantes, E., Holmer, M., 2018. Local regime shifts prevent natural recovery and restoration of lost eelgrass beds along the Swedish West Coast. Estuar. Coast 1–20. https://doi.org/10.1007/s12237-018-0382-y.

Morro Bay National Estuary Program, 2017. Morro bay eelgrass report 2014 to 2016. Retrieved from. http://www.mbnep.org/wp-content/uploads/2014/12/2014-2016-Eelgrass-Report.pdf. (Accessed April 2020).

Ondiviela, B., Losada, I.J., Lara, J.L., Maza, M., Galván, C., Bouma, T.J., van Belzen, J., 2014. The role of seagrasses in coastal protection in a changing climate. Coast. Eng. 87, 158–168. https://doi.org/10.1016/j.coastaleng.2013.11.005.

Orth, R., Carruthers, T., Dennison, W.C., Duarte, C.M., Fourqurean, J.W., Heck, K.L., Hughes, a R., Kendrick, G. a, Kenworthy, J.W., Olyarnik, S., Short, F.T., Waycott, M., Williams, S.L., 2006. A global crisis for seagrass ecosystems, 56, pp. 987–996. https://doi.org/10.1641/0006-3568(2006)56[987:AGCFSE]2.0.CO;2.

Paquier, A.-E., Meulé, S., Anthony, E.J., Larroudé, P., Bernard, G., 2019. Wind-induced hydrodynamic interactions with aquatic vegetation in a fetch-limited setting: implications for coastal sedimentation and protection. Estuar. Coast 42, 688–707. https://doi.org/10.1007/s12237-018-00487-w.

Reidenbach, M.A., Thomas, E.L., 2018. Influence of the seagrass, Zostera marina, on wave attenuation and bed shear stress within a shallow coastal bay. Front. Mar. Sci. 26 https://doi.org/10.3389/fmars.2018.00397.

Shaughnessy, F.J., Gilkerson, W., Black, J.M., Ward, D.H., Petrie, M., 2012. Predicted eelgrass response to sea level rise and its availability to foraging Black Brant in Pacific coast estuaries. Ecol. Appl. 22, 1743–1761. https://doi.org/10.1890/11-1083.1.

Taherkhani, M., Vitousek, S., Barnard, P.L., Frazer, N., Anderson, T.R., Fletcher, C.H., 2020. Sea-level rise exponentially increases coastal flood frequency. Sci. Rep. 10, 6466. https://doi.org/10.1038/s41598-020-62188-4.

Thorne, K., MacDonald, G., Ambrose, R., Buffington, K., Freeman, C., Janousek, C., Brown, L., Holmquist, J., Guntenspergan, G.R., Powelson, K., Barnard, P., Takekawa, J., 2016. Effects of climate change on tidal marshes along a latitudinal gradient in California. USGS Open-File Report 2016-1125. https://doi.org/10.3133/ofr2016112.

Unsworth, R.K.F., McKenzie, L.J., Collier, C.J., Cullen-Unsworth, L.C., Duarte, C.M., Eklöf, J.S., Jarvis, J.C., Jones, B.L., Nordlund, L.M., 2019. Global challenges for seagrass conservation. Ambio 48, 801–815. https://doi.org/10.1007/s13280-018-1115-y.

van Der Heide, T., van Nes, E.H., Geerling, G.W., Smolders, A.J.P., Bouma, T.J., Van Katwijk, M.M., 2007. Positive feedbacks in seagrass ecosystems: implications for success in conservation and restoration. Ecosystems 10, 1311–1322. https://doi.org/10.1007/s10021-007-9099-7.

van der Heide, T., van Nes, E.H., van Katwijk, M.M., Olff, H., Smolders, A.J.P., 2011. Positive feedbacks in seagrass ecosystems - evidence from large-scale empirical data. PLoS One 6, 1–7. https://doi.org/10.1371/journal.pone.0016504.

Vitousek, S., Barnard, P.L., Fletcher, C.H., Frazer, N., Erikson, L., Storlazzi, C.D., 2017a. Doubling of coastal flooding frequency within decades due to sea-level rise. Sci. Rep. 7, 1399. https://doi.org/10.1038/s41598-017-01362-7.

Vitousek, S., Barnard, P.L., Limber, P., Erikson, L., Cole, B., 2017b. A model integrating longshore and cross-shore processes for predicting long-term shoreline response to climate change. J. Geophys. Res.: Earth Surf. 122 (4), 782–806. https://doi.org/10.1002/2016JF004065.

Walter, R.K., Rainville, E.J., O'Leary, J.K., 2018a. Hydrodynamics in a shallow seasonally low-inflow estuary following eelgrass collapse. Estuar. Coast Shelf Sci. 213, 160–175. https://doi.org/10.1016/j.ecss.2018.08.026.

Walter, R.K., Armenta, K.J., Shearer, B., Robbins, I., Steinbeck, J., 2018b. Coastal upwelling seasonality and variability of temperature and chlorophyll in a small coastal embayment. Continent. Shelf Res. 154 https://doi.org/10.1016/j.csr.2018.01.002.

Ward, L., Kemp, M., Walter, B., 1984. The influence of waves and seagrass communities on suspended particulates in an estuarine embayment. Mar. Geol. 58, 85–103.

Waycott, M., Duarte, C.M., Carruthers, T.J.B., Orth, R.J., Dennison, W.C., Olyarnik, S., Calladine, a., Fourqurean, J.W., Heck, K.L., Hughes, a.R., Kendrick, G.a., Kenworthy, W.J., Short, F.T., Williams, S.L., 2009. Accelerating loss of seagrasses across the globe threatens coastal ecosystems. Proc. Natl. Acad. Sci. Unit. States Am. 106, 12377–12381. https://doi.org/10.1073/pnas.0905620106.

Wilson, D.P., 1949. The decline of Zostera marina L. At salcombe and its effects on the shore. J. Mar. Biol. Assoc. U. K. 28, 395–412. https://doi.org/10.1017/S0025315400023298.