The Geomorphic Impact of Mangrove Encroachment in an Australian Salt Marsh

Daniel J. Coleman[1], Kerrylee Rogers[2], D. Reide Corbett[3], Christopher J. Owers[4], Matthew L. Kirwan[1]

1. Virginia Institute of Marine Science
2. University of Wollongong
3. East Carolina University
4. Aberystwyth University

Abstract

Mangroves are encroaching into salt marshes throughout the world as a result of environmental change. Previous studies suggest mangroves trap sediment more efficiently than adjacent salt marshes, providing mangroves greater capacity to adapt to sea level rise; this may occur by displacing salt marshes. However, sediment transport in adjacent marsh-mangrove systems and its role in mangrove encroachment upon salt marsh remain poorly understood. Here we directly test the hypothesis that mangroves reduce the ability of adjacent marsh to adjust to sea level rise by measuring sediment transport across salt marsh platforms, with and without six meters of fringing mangroves at the tidal creek edge. We find that salt marshes and mangroves have equivalent sediment trapping efficiencies along the wetland edge. Suspended sediment concentrations, mass accumulation rates, and long-term accretion rates are not lower in salt marshes landward of mangroves than salt marshes without fringing mangroves. Therefore, our work suggests that a

relatively narrow zone of mangroves does not impact salt marsh accretion, and activities that limit mangrove encroachment into salt marsh, such as removal of seedlings, will not improve the capacity of salt marsh to trap sediments.

1. Introduction

Mangrove swamps and salt marshes are valuable and vulnerable ecosystems that occupy intertidal environments around the world. They offer similar ecosystem services such as storm surge protection, carbon sequestration, and nursery habitat provision (Kelleway et al., 2017; Himes-Cornell et al., 2018; Lefcheck et al., 2019). Both coastal wetland communities are threatened by human impacts, especially reduced sediment supply and anthropogenic sea level rise. One key difference between these wetland communities is the latitudinal ranges they occupy, with mangroves dominating in the tropics and salt marshes dominating coastlines in temperate zones. Where these communities overlap around the world, marshes are often displaced by mangroves, which themselves are range-limited by physical factors (Kangas and Lugo, 1990; Guo et al., 2013; Saintilan et al., 2014).

Climate change tends to promote mangrove encroachment into salt marsh (Saintilan and Wilton, 2001; Krauss et al., 2011; Osland et al., 2013; Saintilan et al., 2014; Rogers and Krauss, 2019), however the drivers of mangrove encroachment are regionally variable. For example, the predominant poleward expansion of mangroves in the USA has been attributed to a reduction in the number of winter freeze events (Osland et al., 2013; Cavanaugh et al., 2014). However, Australia has a more cold-tolerant mangrove species distributed across the entire mainland coast (Duke 2006; Rogers and Krauss, 2019). In addition, mangroves in Australia typically occupy more seaward locations and lower elevations than marshes (Clarke and Hannon, 1967; Rogers and Krauss, 2019;

Owers et al. 2020). For example, a previous study in New South Wales (NSW) Australia found shrubby mangroves occupied an elevation interquartile range of 0.188 m to 0.525 m compared to a salt marsh elevation interquartile range of 0.572 m to 0.737 m, relative to the Australian Height Datum (Owers et al. 2020). Rising sea level can convert land suitable for salt marsh vegetation into habitat for mangroves if the elevation of salt marsh does not increase at a rate proportional to sea level rise (Rogers et al., 2005; Rogers and Krauss, 2019). As a result, the primary direction of encroachment is landward rather than poleward (Saintilan and Williams, 1999; Rogers and Krauss, 2019).

In some ecosystems, certain species are known to facilitate the displacement of competitors. This conspecific or self-facilitation can occur through many different mechanisms such as allelopathy (Rice 2012), shading (Kangas and Lugo, 1990), and altering fire regimes (Brooks et al., 2004). In the mangrove-salt marsh system of Australia, it remains unclear if the presence of mangroves promotes further encroachment. A mechanism of mangrove self-facilitation in this system could be sediment trapping. If creek-adjacent mangroves trap sediment such that interior marshes receive less sediment, they may be less capable of maintaining elevation relative to rising sea level. The marsh may then lose elevation with respect to the tidal frame, which could promote landward mangrove expansion. The expansion of the fringing mangrove into salt marsh habitat could then trap more sediment creating a positive feedback that results in prolific mangrove expansion into marsh communities. Mangrove expansion has been shown to be more rapid in areas with lower marsh accretion, which is consistent with this proposed mechanism (Rogers et al., 2006).

Removal and containment of mangrove establishment has been proposed to protect salt marsh, as well as improve recreation and property value (Harty 2009). In the Hunter Estuary in NSW, Australia, mangroves have been removed to restore salt marsh habitat for migratory shorebirds (Reid 2019). Temperate zone salt marsh is identified as an ecologically threatened community under both federal and NSW state legislation (Commonwealth Environmental Protection and Biodiversity Conservation Act 1999, NSW Threatened Species Conservation Act 1995), and sea level rise has been identified as a key threatening process (Rogers et al., 2016). However, the geomorphological and ecological impact of mangrove removal on salt marshes has yet to be established. Previous research suggests that mangroves are more efficient at capturing sediment than marshes, however this finding is inconsistent and confounded by differences in elevation across sites (Rogers et al., 2005; Perry et al., 2009; Lovelock et al. 2014; Kelleway et al., 2017). If superior sediment trapping of mangroves is attributable to vegetation structure, then removing mangroves may allow for increased sediment supply to the landward salt marshes. However, if observed differences in sediment trapping are a result of relative position in the tidal frame, then removing mangroves would not be expected to affect salt marsh sediment supply. Further investigation is necessary before such an ambitious and potentially ecologically-harmful project is undertaken.

In this study, we aim to quantify how mangroves influence sediment transport to a landward salt marsh by comparing two locations within a wetland system. One location has creek-adjacent mangroves with landward salt marsh and one has salt marsh abutting the creek edge, which represents a rare occurrence in the region. We hypothesize that mangroves will decrease suspended sediment concentration (SSC) more than salt marsh plants, leading to reduced short and long-term vertical accretion of landward salt marsh. It is anticipated that this study will provide crucial information to support decision making and intervention both now and in the future as mangroves continue to encroach upon salt marshes.

2. Methods

2.1 Study Site

We measured key indicators of sediment transport along two transects in Currambene Creek, which enters Jervis Bay in NSW, Australia (Figure 1). The estuary has a semi-diurnal tidal range of approximately 2 meters and supports temperate coastal wetland communities characteristic of southeast Australia (Clarke, 1993; Owers et al. 2016). The transects were positioned shore-normal and progressed from the tidal creek to approximately 15 m into the marsh interior, with and without fringing mangroves. The transect that traversed both mangrove and salt marsh, henceforth termed ‘mixed transect’, was approximately 650 m downstream of the marsh-only transect. The mixed transect had a narrow fringe of *Aegiceras corniculatum* mangroves approximately <2 m tall and 6 m in width along the creek, and nearby *Avicennia marina* mangroves, with extensive *Sporobolus virginicus* salt marsh landward. The marsh transect only traversed salt marsh vegetation and featured *S. virginicus* adjacent to the tidal creek. We conducted a topographic survey with a real-time kinematic GPS (8 mm horizontal precision and 15 mm vertical precision) to ensure each site had a similar elevation range (Figure 2). The mixed transect elevations decreased gradually to the creek whereas the marsh transect—which was located on the cut bank of a meander of Currambene Creek—had a scarp. The elevation range on the wetland surface itself was similar for both transects (0.325 m to 0.483 m for the mixed transect and 0.410 m to 0.462 m for the marsh transect, relative to the Australian Height Datum).

2.2 Suspended Sediment Concentrations

Optical back scatter turbidity sensors (RBR brand) were deployed along the two transects to quantify sediment transport in the presence and absence of fringing mangroves. On each transect, one sensor was located 30 cm above the substrate within the tidal creek, and four sensors were located 7cm above the bed on the marsh or mangrove surface (Figure 1). Sensors were deployed on the mixed transect so that two were located within the mangrove vegetation and two were in the salt marsh vegetation. Sensors along the marsh transect were located at distances from the creek that correspond to the distances along the mixed transect. All sensors recorded turbidity every 15 minutes from the end of July/beginning of August to mid-September 2018. This record was divided into flooding and ebbing tides based on atmospheric-corrected pressure data from the creek sensors (atmospheric data from Australian Bureau of Meteorology). The pressure record was also used to remove any data points in which a given sensor was not flooded. We also removed data points in which a sensor was fouled or obstructed following Ganju et al. (2005). Turbidity was calibrated to SSC via *in situ* sampling and laboratory calibration following Coleman and Kirwan (2019), resulting in the equation SSC (mg/L) = 0.6421 * Sensor Turbidity (NTU) ($n=20$, $R^2=0.9146$, $p<0.01$).

2.3 Vegetation Biomass

To quantify plant biomass of both marsh and mangrove species, we utilized non-destructive measurement techniques that have shown to be statistically equivalent to harvest measurements (Owers et al. 2018a; Owers et al. 2018b). Marsh biomass was measured via terrestrial laser scanning (Owers et al. 2018a) in a 1 m^2 plot at each sensor location and at one additional plot located 13 m farther landward. The calculation of biomass from the terrestrial laser scans depends on statistical relationships previously determined for this site (Owers et al., 2018a). For mangrove biomass, we measured height, crown area, and stem circumference for all branches greater than 5 cm circumference for all mangrove individuals within a 25 m^2 plot centered on the transect. These measurements were used to calculate biomass using allometric equations generated for *A. corniculatum* and *A. marina* at this study site (Owers et al., 2018b). To compare between discrete, individual mangroves and plot-scale marsh biomass, we standardized mangrove biomass by calculating biomass per square meter. This results in a single value of mangrove biomass per square meter without an associated standard deviation.

2.4 Mass Accumulation and Trapping Efficiency

We deployed filter paper sediment traps to measure short-term mass accumulation rates. The sediment traps consisted of pre-combusted, pre-weighed 90 mm glass microfiber filter paper fixed to square ceramic tiles that were staked to the marsh surface. Five replicate traps were deployed at each sensor on the wetland surface at the end of July/beginning of August and collected in mid-September (Figure 1). We then calculated the mass of the accumulated sediment divided by the deployment length standardized to the filter paper area to determine a mass accumulation rate (mg/cm^2/day). Additionally, we calculated sediment trapping efficiency ((g/cm^2)/(mg/L)) as the mass accumulation per unit area divided by the mean SSC at that corresponding sensor.

For both mass accumulation rate and trapping efficiency, we constructed linear mixed effect models with fixed effect variables of transect type (mixed or marsh-only) and distance from the shoreline and random variables of vegetation biomass and sediment trap replicate. Elevation was collinear with distance and therefore not included as a separate variable within the model. This model can be used to determine how mass accumulation and trapping efficiency vary with distance and if the relationship with distance is different between the transects.

No transformations of the data were needed, and the analysis was performed in MATLAB. We determined which model was the best via the model-quality estimator Akaike information criterion (AIC) and a parsimony analysis. We can therefore test our hypothesis that more sediment is retained within the fringing mangroves and less penetrates into the landward marsh compared to the marsh-only transect. We would expect the fringing mangroves of the mixed transect to have the highest mass accumulation rate and trapping efficiency while the landward salt marsh of the mixed transect to have the lowest mass accumulation rate.

2.5 Sediment Properties and Accretion Rates

To measure long-term accretion rates and sediment properties, we collected two sediment cores from each site. One core was located 1 m from the shoreline and the other 7 m from the shoreline, corresponding with sensor locations (Figure 1). Cores were sectioned into 1 cm increments over the first 20 cm. We calculated bulk density for each section by determining the mass of a 1 cm diameter x 1 cm height cylindrical subsample dried at 100°C until a constant mass was achieved. Dry subsamples were then ground and combusted at 500°C for 8 hours to determine loss on ignition for organic matter content in accordance with Ball (1964). We calculated percent sand, silt, and clay of each 1 cm increment in the top 20 cm using a Malvern Mastersizer 2000. Finally, we calculated accretion rates using radioisotope geochronology. Dating recent sediments usually relies on the determination of the vertical distribution of unsupported or 'excess' ^{210}Pb (half-life 22.3 years), a naturally occurring fallout radionuclide, which then allows ages to be ascribed to sedimentary layers based on the known decay rate of ^{210}Pb (Appleby and Oldfield, 1992). Briefly, total ^{210}Pb was measured by alpha spectroscopy following Nittrouer et al. (1979) where ca. 1.5 g of sediment was spiked with ^{209}Po followed by partial digestion with 8 N nitric acid (HNO3) by microwave heating. We electroplated ^{209}Po and ^{210}Po onto nickel planchets in a dilute acid solution. The supported ^{210}Pb activity for this core was assumed to be equal to the uniform background activity found at depth (Nittrouer et al., 1979). The ^{210}Pb-derived accretion rates are based on the constant rate of supply (CRS) model (Appleby and Oldfield, 1992; Corbett and Walsh, 2015). The average accretion rates calculated over the past 50 years for each core were then statistically compared to one another.

3. Results

3.1 Suspended Sediment Concentrations

The spatial pattern of average SSC for the entire record and when flood or ebb tides were isolated was similar between the two transects, despite different wetland edge morphology (Figure 3). The concentrations were low with an average of 1.1 mg/L along the mixed transect and 0.9 mg/L along the marsh transect. Concentrations were generally highest in the creek and decreased rapidly with distance into the wetland (Figure 3). The notable exception to this pattern was a substantial increase in SSC at the landward sensor of the fringing mangroves of the mixed site. This increase did not appear to affect the amount of sediment reaching the marsh. On the flooding tide, the first marsh sensor of the mixed transect had an average SSC that was 43% less than the creek sensor, which translates to a 43% decrease in the SSC of water passing through the fringing mangroves. Over the corresponding distance along the marsh transect, there was a 74% decrease in SSC. Spatial patterns are similar on flood tides (b) and ebb tides (c).

3.2 Vegetation Biomass

Vegetation biomass was similar between transects. The average biomass of mangrove and marsh plants at the mixed transect (4.84 kg/m^2 ± 2.40) was not significantly different than the average biomass of the marsh-only transect (4.70 kg/m^2 ± 2.00; Student's t-test, p=0.92). Similarly, the average biomass of the marsh portion of the mixed transect (4.31 kg/m^2 ± 2.9) was not significantly different than the average biomass at the corresponding locations in the marsh-only transect (5.34 kg/m^2 ± 1.35; Student's t-test, p=0.60). The mangrove biomass was 5.91 kg/m^2 while the marsh edge biomass was 3.96 ± 2.39 kg/m^2.

3.3 Mass Accumulation and Trapping Efficiency

Mass accumulation rate and sediment trapping efficiency increased with distance from the shore at both the mixed and marsh-only transects (Figure 4). Mass accumulation rate ranged from 0.07 to 0.21 (mg/cm^2/day) and sediment trapping efficiency ranged from 2.24 to 12.3 (g/cm^2)/(mg/L). Based on the model

AIC and parsimony, the best linear mixed effect model for mass accumulation was Mass Accumulation~ Distance + (1|Trap Replicate) and for trapping efficiency was Trapping Efficiency~ Distance + (1|Trap Replicate). Distance from the shore was a significant predictor for both mass accumulation rate and sediment trapping efficiency (regression coefficient $\beta=0.0085 \pm 0.0034$, $p<0.05$; $\beta=0.72 \pm 0.19$, $p<0.001$), whereas transect was not (Figure 4). That is, for a given distance from the shoreline, mass accumulation rate and sediment trapping efficiency are not different between the mixed transect and the marsh transect ($\beta=-0.021 \pm 0.024$, $p=0.39$; $\beta=1.61 \pm 1.50$, $p=0.29$, Figure 4).

3.4 Sediment Properties and Accretion Rates

Sediment properties were generally consistent between the top 20 cm of cores from all locations (Figure 5). The site with the lowest average bulk density of the top 20 cm was located in the fringing mangroves of the mixed transect (1.11 g/cm^3, range of 0.55 g/cm^3, Figure 5) and the site with the highest average bulk density was 7 m from the shore at the marsh-only transect (1.28 g/cm^3, range of 0.65 g/cm^3). Inter-site variability of average bulk density was less than the variability in bulk density with depth in a given core. Average organic matter content in the top 20 cm was very low in all cores, with no core having greater than 5% by mass. The cores from the fringing mangroves had the coarsest grain size (54% sand, range of 24%) and the corresponding location in the marsh site was the finest (34% sand, range of 31%, Figure 5). The interior marshes had an average sand content of 46% (range of 18%) and 40% (range of 32%) for the mixed and marsh-only transects, respectively (Figure 5).

Long-term accretion rates were higher along the mixed transect compared to the marsh-only transect and in the interior compared to the edge. The accretion rates averaged over the last 50 years for all cores were significantly different from one another (ANOVA, $F(3,29)=5.59$, $p<0.05$). The mangrove portion of the mixed transect accreted significantly faster than the marsh edge at the marsh-only transect (2.1 mm/yr compared to 1.1 mm/yr, Student's t-test, $p<0.05$). Similarly, the marsh portion of the mixed transect accreted significantly faster than the marsh interior at the marsh-only transect (2.6 mm/yr compared to 1.6 mm/yr, Student's t-test, $p<0.05$). Within a single transect, the edge and the interior did not have statistically different accretion rates (Student's t-test, $p=0.21$ and $p=0.25$ for the mixed and marsh-only transects, respectively). Accretion rates generally increased through time; rates in the most recent 20 years are faster than any other time in the past 100 at all sites except the marsh interior of the marsh-only transect (Figure 6).

Discussion

The presence of mangroves does not appear to alter sediment supply to the landward marshes. The percentage of sediment passing through the fringing mangroves and into the marsh (57%) was actually greater than the percentage that passed through the corresponding location at the marsh-only transect (26%; Figure 3). Similarly, the mass accumulation rate and long-term accretion rate of the marsh at the mixed transect was not significantly lower than the corresponding location at the marsh-only transect (Figure 4).

There was a noticeable increase in SSC in the landward portion of the mangrove fringe. However, this increase was present in both flood and ebb tides and the mass accumulation rate at that sensor was not elevated. This suggests the increase is not indicative of significant sediment deposition and instead represents sediment simply passing through a localized, more dynamic sedimentary environment (Figure 3 and 4). The sediment concentrations, accumulation, and accretion rates are extremely low in this system. It is possible that our inability to detect a difference between the marsh components between the two transects may be partially attributable to these low values. Additionally, it is possible that a wider fringe of mangroves could exaggerate differences in sediment supply to the marsh components of the transect. However, all three of the sediment-based metrics were actually higher in the marsh component of the mixed transect than the corresponding location at the marsh-only transect. These results suggest that mangrove do not promote further encroachment into salt marsh via limiting sediment transport to interior marshes.

Mangrove encroachment is an international phenomenon with numerous regionally-specific drivers (Saintilan and Wilton, 2001; Krauss et al., 2011; Osland et al., 2013; Cavanaugh et al., 2014; Saintilan et al., 2014; Rogers and Krauss, 2019). In regions where changing temperatures drive mangrove encroachment, such as North America (Osland et al., 2013; Cavanaugh et al., 2014), we would not expect the sediment dynamics explored in this work to be a mechanism of encroachment. More broadly though, our work suggests mangroves and salt marsh species have similar abilities to capture sediment. This study is therefore informative in the implications of temperature-driven encroachment, suggesting that the change in vegetation species will not result in a major change in sediment dynamics. Our study is of course most relevant to regions where sea level rise is the main driver of mangrove encroachment. Even in these regions, other factors can play a role in encroachment, such as rainfall (Duke et al., 2019; Rogers and Krauss, 2019), hysteresis (Duke et al. 2019), and nutrients (Lovelock et al. 2007). These additional drivers can alter the elevation of the mangrove-marsh ecotone. While it is possible that sediment dynamics may influence encroachment more in other environments with similar vegetation zonation, we found no indication that mangrove fringes prevent more sediment from reaching interior locations than corresponding widths of salt marsh.

While previous work suggests that mangroves have higher accretion rates than salt marsh and may be more resilient to sea level rise, the underlying mechanism remains unclear (Rogers et al., 2005; Lovelock et al. 2014; Kelleway et al., 2017). Some of the accretion difference may be attributable to the position of mangroves lower in the tidal frame than salt marsh, particularly in Australia (Kelleway et al., 2017). Wetlands lower in the tidal frame tend to have greater sediment accumulation rates because they are inundated for longer durations (Friedrichs and Perry, 2001; FitzGerald et al., 2008; Kirwan et al., 2016). For example, mangroves in North America are not consistently lower in elevation than marshes and do not have consistently higher accretion rates (Perry et al., 2009; Bianchi et al., 2013; McKee and Vervaeke, 2017).

In our study, mangroves occurred at an elevation range that encompassed the elevation range of the salt marsh (Figure 2), and mass accumulation rates were not different between mangroves and salt marshes at similar distances from the creek edge (Figure 4). This suggests that previous differences in accumulation rates between vegetation types may instead depend on relative elevation rather than vegetation type. However, there were differences in long-term accretion rates between vegetation types with higher rates in the mangroves than marshes at corresponding locations (Figure 6a). The greater surface elevation gain in mangrove may also be influenced by the greater contribution of mangrove organic matter to substrates than salt marsh (Rogers et al., 2005; Kelleway et al., 2017), and this contribution may not be detectable in surface accretion. Mangroves may therefore be more capable than salt marsh at building elevation to keep pace with rising sea level due to belowground rather than aboveground processes. The differential response to sea level rise caused by belowground elevation building could therefore be a driving factor of mangrove encroachment.

Mangrove vegetation may also attenuate waves to a greater extent than salt marsh vegetation, however this remains uncertain (Gedan et al., 2011; Kelleway et al., 2017). Direct comparisons of wave attenuation between salt marsh and mangroves are limited and often do not account for other environmental factors (Kelleway et al., 2017). However, previous research has shown vegetation composition and structure are important factors in the efficiency of dissipating energy from incoming flows (Montgomery et al. 2018). While more research is needed, the greater aboveground biomass, structural complexity, and rigidity of mangroves could theoretically make them more capable of wave attenuation than salt marsh assemblages (Kelleway et al., 2017). Mangroves may therefore protect the landward marshes and aid the stability of the entire mixed marsh-mangrove system.

Our finding that fringing mangroves do not disrupt sediment delivery to interior marshes is inconsistent with mangrove removal as an effective salt marsh management strategy. Removal of mangroves have been considered by various local and state governments in Australia due to concerns that sea-level-driven mangrove encroachment will displace ecologically vulnerable salt marshes (Harty 2009). For example, several stakeholders have proposed mangrove removal for Lake Illawarra (NSW, Australia) if it is shown that mangroves have a detrimental impact on salt marsh (Lake Illawarra Coastal Management Plan, 2019; Rogers personal observations). The efficacy of removing mangroves to preserve salt marsh has not been demonstrated. In our study, the presence of mangroves had little effect on rates of sediment accumulation in adjacent marshes and will be unlikely to have a detectable influence on the capacity of salt marsh to adapt to sea level rise.

Our findings are consistent with previous studies that suggest mangroves have greater capacity than marshes for sustained surface accretion given long-term sea level rise (Rogers et al., 2005; Saintilan and Rogers, 2006). Given the higher rates of long-term accretion (Figure 6a), the potential of mangroves to attenuate waves better than marshes (Kelleway et al., 2017), and the greater contribution to belowground organic matter (Rogers et al. 2005), the removal of mangroves may have a net negative impact on the ability of the wetland system to respond to sea level rise. Mangrove encroachment has been described as a symptom of other environmental change processes (Harty 2009). Mangrove removal is an ineffective management strategy that does not address the underlying cause of vegetation shifts (Harty 2009). There is an effective prioritization of salt marsh management over mangrove management (Harty 2009), however both vegetation communities offer valuable ecosystem services on their own (Kelleway et al., 2017; Himes-Cornell et al., 2018; Lefcheck et al., 2019) and synergistically (Saintilan et al. 2007). Other management strategies, such as limiting development into coastal wetlands or allowing for upland migration, will likely be more effective at protecting both salt marsh and mangroves (Evans and Williams, 2001; Kirwan et al., 2016; Schieder et al., 2018).

Conclusion

We found that fringing mangroves had no negative effects on sediment supply, mass accumulation rate, or long-term accretion rates of a landward salt marsh. These results suggest that mangrove expansion does not lead to sediment deficits in adjacent salt marshes, and that removing mangroves would not aid salt marsh capacity to respond to sea level rise. Previous work suggests mangroves may be more capable than marshes at responding to sea level changes and reducing erosion from waves and storms. Thus, our work suggests that the removal of mangroves to protect ecologically threatened marshes may have a negative impact on the marsh and wetland system as a whole.

Acknowledgement

This project was funded by the National Science Foundation (NSF) Graduate Research Fellowship Program, Graduate Research Opportunities Worldwide and the University of Wollongong. Kirwan was supported by NSF LTER #1832221, NSF EAR-CAREER #1654374, NSF EAR-GLD #1529245, NSF OCE-SEES #1426981. The authors would like to thank E. Asbridge, K. Lal, L. Mogensen, M. Goddard, J. Kelleway, E. Coleman, and S. Coleman for assistance in the field. This is contribution number XXXXX of the Virginia Institute of Marine Science.

References

Appleby, P.G., Oldfield, F., 1992. Application of 210Pb to sedimentation studies. In: Ivanovich, M., Harmon, R.S. (Eds.), Uranium Series Disequilibrium. Oxford University Press, Oxford, pp. 731-778.

Bianchi, T. S., Allison, M. A., Zhao, J., Li, X., Comeaux, R. S., Feagin, R. A., & Kulawardhana, R. W. (2013). Historical reconstruction of mangrove expansion in the Gulf of Mexico: linking climate change with carbon sequestration in coastal wetlands. *Estuarine, Coastal and Shelf Science*, *119*, 7-16.

Brooks, M. L., D'antonio, C. M., Richardson, D. M., Grace, J. B., Keeley, J. E., DiTomaso, J. M., ... & Pyke, D. (2004). Effects of invasive alien plants on fire regimes. *BioScience*, *54*(7), 677-688.

Cavanaugh, K. C., Kellner, J. R., Forde, A. J., Gruner, D. S., Parker, J. D., Rodriguez, W., & Feller, I. C. (2014). Poleward expansion of mangroves is a threshold response to decreased frequency of extreme cold events. Proceedings of the National Academy of Sciences, 111(2), 723-727.

Clarke, L. D., & Hannon, N. J. (1967). The mangrove swamp and salt marsh communities of the Sydney district: I. Vegetation, soils and climate. *The Journal of Ecology*, 753-771.

Clarke, P.J. (1993). Mangrove, saltmarsh and peripheral vegetation of Jervis Bay. *Cunninghamia 3*(1): 231-253.

Coleman, D.J., & Kirwan, M.L., (2019). The effect of a small vegetation dieback event on salt marsh sediment transport. *Earth Surface Processes and Landforms* 44(4), 944-952, https://doi.org/10.1002/esp.4547

Corbett, D. R., and J. P. Walsh. 2015. ^{210}Lead and ^{137}Cesium Establishing a chronology for the last century. In I. Shennan, A. J. Long, and B. P. Horton (Ed.) Handbook of sea-level research. John Wiley & Sons, Inc. Pp. 361-372.

Duke, N. C. (2006). *Australia's mangroves: the authoritative guide to Australia's mangrove plants*. MER.

Duke, N. C., Field, C., Mackenzie, J. R., Meynecke, J. O., Wood, A. (2019). Rainfall and its possible hysteresis effect on the proportional cover of tropical tidal-wetland mangroves and saltmarsh-saltpans. *Marine nd Freshwater Resources* doi.org/10.1071/MF18321

Evans M. J., and R. J. Williams. (2001). Historical distribution of estuarine wetlands at Kurnell Peninsula, Botany Bay. *Wetlands (Australia)* 19(2):61–71.

FitzGerald, D. M., Fenster, M. S., Argow, B. A., & Buynevich, I. V. (2008). Coastal impacts due to sea-level rise. Annual Review of Earth and Planetary Sciences, 36.

Friedrichs, C. T., & Perry, J. E., (2001). Tidal salt marsh morphodynamics: A synthesis Journal of Coastal Research 27, 7-37. https://www.jstor.org/stable/25736162

Ganju, N.K., Schoellhamer, D.H. & Bergamaschi, B.A., (2005). Suspended sediment fluxes in a tidal wetland: Measurement, controlling factors, and error analysis. *Estuaries 28*(6), 812-822.

Gedan, K. B., Kirwan, M. L., Wolanski, E., Barbier, E. B., & Silliman, B. R. (2011). The present and future role of coastal wetland vegetation in protecting shorelines: answering recent challenges to the paradigm. *Climatic change*, *106*(1), 7-29.

Guo, H., Zhang, Y., Lan, Z., & Pennings, S. C. (2013). Biotic interactions mediate the expansion of black mangrove (A vicennia germinans) into salt marshes under climate change. *Global Change Biology*, *19*(9), 2765-2774.

Harty, C. (2004). Planning strategies for mangrove and saltmarsh changes in Southeast Australia. *Coastal Management*, *32*(4), 405-415.

Himes-Cornell, A., Grose, S. O., & Pendleton, L. (2018). Mangrove ecosystem service values and methodological approaches to valuation: Where do we stand?. *Frontiers in Marine Science*, *5*, 376.

Kangas, P. C., & Lugo, A. E. (1990). The distribution of mangroves and saltmarsh in Florida. *Tropical Ecology*, *31*(1), 32-39.

Kelleway, J. J., Cavanaugh, K., Rogers, K., Feller, I. C., Ens, E., Doughty, C., & Saintilan, N. (2017). Review of the ecosystem service implications of mangrove encroachment into salt marshes. *Global Change Biology*, *23*(10), 3967-3983.

Kirwan, M. L., Temmerman, S., Skeehan, E. E., Guntenspergen, G. R., & Fagherazzi, S. (2016). Overestimation of marsh vulnerability to sea level rise. *Nature Climate Change*, *6*(3), 253-260.

Krauss, K. W., From, A. S., Doyle, T. W., Doyle, T. J., & Barry, M. J. (2011). Sea-level rise and landscape change influence mangrove encroachment onto marsh in the Ten Thousand Islands region of Florida, USA. *Journal of Coastal Conservation*, *15*(4), 629-638.

Lake Illawarra Coastal Management Plan Draft Report (2019). New South Wales Office of Environment and Heritage. https://our.wollongong.nsw.gov.au/lake-illawarra-coastal-zone-management-plan1

Lefcheck, J. S., Hughes, B. B., Johnson, A. J., Pfirrmann, B. W., Rasher, D. B., Smyth, A. R., Williams, B. L., Beck, M. W., & Orth, R. J. (2019). Are coastal habitats important nurseries? A meta-analysis. *Conservation Letters*, *12*(4), e12645.

Lovelock, C. E., Feller, I. C., Ellis, J., Schwarz, A. M., Hancock, N., Nichols, P., Sorell, B., (2007). Mangrove growth in New Zealand estuaries: the role of nutrient enrichment at sites with contrasting rates of sedimentation. *Oecologia 153* 633-641.

Lovelock, C. E., Adame, M. F., Bennion, V., Hayes, M., O'Mara, J., Reef, R., & Santini, N. S. (2014). Contemporary rates of carbon sequestration through vertical accretion of sediments in mangrove forests and saltmarshes of South East Queensland, Australia. *Estuaries and coasts*, *37*(3), 763-771.

McKee, K. L., Vervaeke, W. C., (2017). Will fluctuations in salt marsh-mangrove dominance alter vulnerability of a subtropical wetland to sea-level rise? *Global Change Biology 24*(3), https://doi.org/10.1111/gcb.13945

Montgomery, J. M., Bryan, K. R., Horstman, E. M., & Mullarney, J. C. (2018). Attenuation of tides and surges by mangroves: contrasting case studies from New Zealand. *Water*, *10*(9), 1119.

Nittrouer, C.A., Sternberg, R.W., Carpenter, R. and Bennett, J.T., 1979. The use of 210Pb geochronology as a sedimentological tool: application to the Washington Continental Shelf. Marine Geology 31, 297–316.

Osland, M. J., Enwright, N., Day, R. H., & Doyle, T. W. (2013). Winter climate change and coastal wetland foundation species: salt marshes vs. mangrove forests in the southeastern United States. *Global change biology*, *19*(5), 1482-1494.

Owers, C. J., Rogers, K., Mazumder, D., & Woodroffe, C. D. (2016). Spatial variation in carbon storage: A case study for Currambene Creek, NSW, Australia. *Journal of Coastal Research*, *75*(sp1), 1297-1301.

Owers, C. J., Rogers, K., & Woodroffe, C. D. (2018a). Terrestrial laser scanning to quantify above-ground biomass of structurally complex coastal wetland vegetation. *Estuarine, Coastal and Shelf Science*, *204*, 164-176.

Owers, C. J., Rogers, K., & Woodroffe, C. D. (2018b). Spatial variation of above-ground carbon storage in temperate coastal wetlands. *Estuarine, Coastal and Shelf Science*, *210*, 55-67.

Owers, C. J., Rogers, K., Mazumder, D., Woodroffe, C. D. (2020). Temperate coastal wetland near-surface carbon storage: Spatial Patterns and variability. *Estuarine, Coastal and Shelf Science, 235,* 106584

Perry, C. L., & Mendelssohn, I. A. (2009). Ecosystem effects of expanding populations of *Avicennia germinans* in a Louisiana salt marsh. *Wetlands*, *29*(1), 396-406.

Reid, P. (2019). Changes in wetland use by shorebirds following mangrove removal, Area E, Ash Island, New South Wales. *The Whistler* 13, 62-68

Rice, E. L. (2012). *Allelopathy*. Academic press.

Rogers, K., Boon, P. I., Branigan S., Duke, N. C., Field, C. D., Fitzsimons, J. A., Kirkman, H., Mackenzie, J. R., Saintilan, N., (2016). The state of legislation and policy protecting Australia's mangrove and salt marsh and their ecosystem services. *Marine Policy 72* 139-155.

Rogers, K., & Krauss, K. W. (2019). Moving from generalisations to specificity about mangrove–saltmarsh dynamics. *Wetlands*, *39*(6), 1155-1178.

Rogers, K., Saintilan, N., & Heijnis, H. (2005). Mangrove encroachment of salt marsh in Western Port Bay, Victoria: the role of sedimentation, subsidence, and sea level rise. *Estuaries*, *28*(4), 551-559.

Saintilan, N., Hossain, K., & Mazumder, D. (2007). Linkages between seagrass, mangrove and saltmarsh as fish habitat in the Botany Bay estuary, New South Wales. *Wetlands Ecology and Management*, *15*(4), 277-286.

Saintilan, N., & Rogers, K. (2006). Coastal wetland elevation trends in southeast Australia. In *Catchments to Coast. Society of Wetland Scientists 27th International Conference* (pp. 42-54).

Saintilan, N., & Williams, R. J. (1999). Mangrove transgression into saltmarsh environments in south-east Australia. *Global Ecology and Biogeography*, *8*(2), 117-124.

Saintilan, N., Wilson, N. C., Rogers, K., Rajkaran, A., & Krauss, K. W. (2014). Mangrove expansion and salt marsh decline at mangrove poleward limits. *Global change biology*, *20*(1), 147-157.

Saintilan, N., & Wilton, K. (2001). Changes in the distribution of mangroves and saltmarshes in Jervis Bay, Australia. *Wetlands Ecology and Management*, *9*(5), 409-420.

Schieder, N. W., Walters, D. C., & Kirwan, M. L. (2018). Massive upland to wetland conversion compensated for historical marsh loss in Chesapeake Bay, USA. *Estuaries and Coasts*, *41*(4), 940-951.

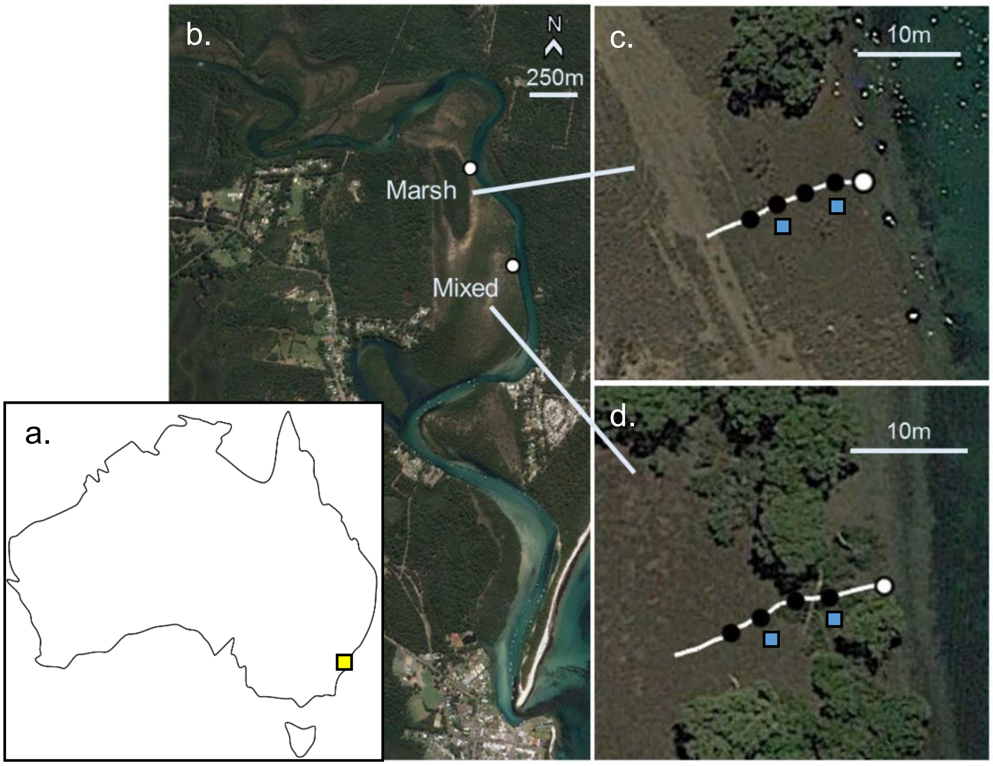

Figure 1. (a.) Site maps showing the location of Currambene Creek (yellow box) on the map of Australia (b.) and each transect with respect to Currambene Creek (white circle). (c.) At the marsh-only transect (d.) and the mixed transect , black circles represent locations of sediment tiles and the sensors on the wetland surface. The white point represents the sensor in the creek and blue squares represent the core locations. The white line indicates the GPS transects.

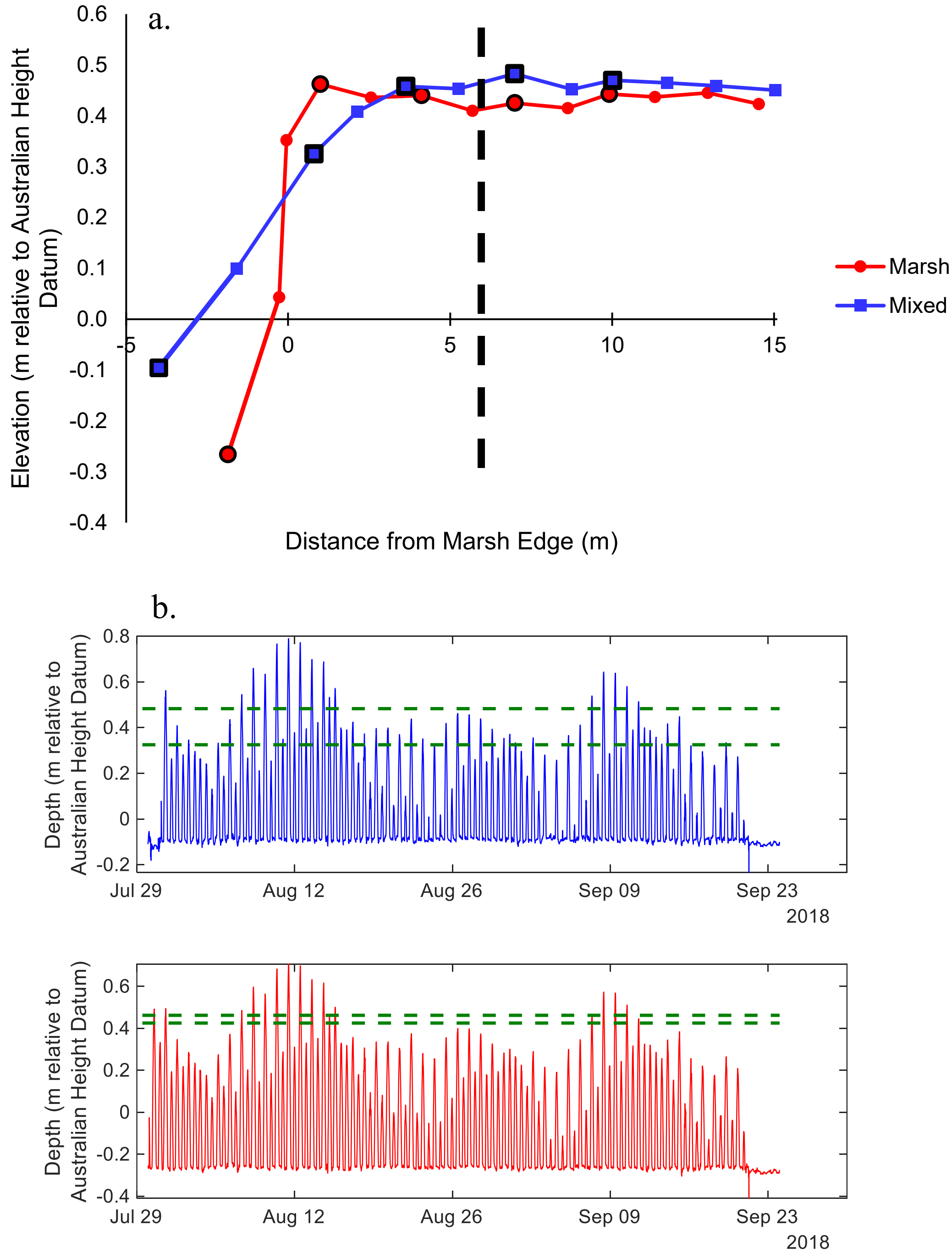

Figure 2. (a.) Elevation profiles for the mixed site (blue) and marsh-only site (red), where points outlined in black indicate sensor locations. The black dashed line indicates the boundary between mangroves and marsh at the mixed transect. (b.) Time series of water level measured in the creek sensor. Horizontal, green dashed lines indicate the minimum and maximum elevations of the sensors on the wetland platform.

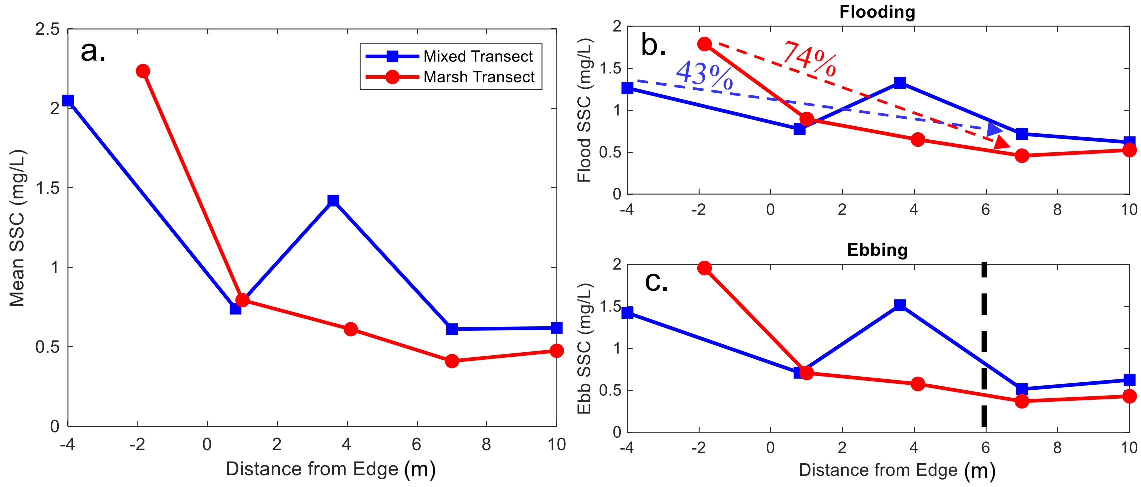

Figure 3. Spatial patterns of SSC across the wetland platform, where distance from edge refers to the distance from a tidal creek. (a) SSC averaged over the entire period of record declines with distance except for localized high concentrations within the fringing mangroves. (b) Average SSC for flooding tides only, where the colored dashed line indicates the percent decrease in creek SSC over the width of the fringing mangroves and (c) Average SSC for ebbing tides only, where the black dashed line indicates the boundary between mangroves and marsh at the mixed transect.

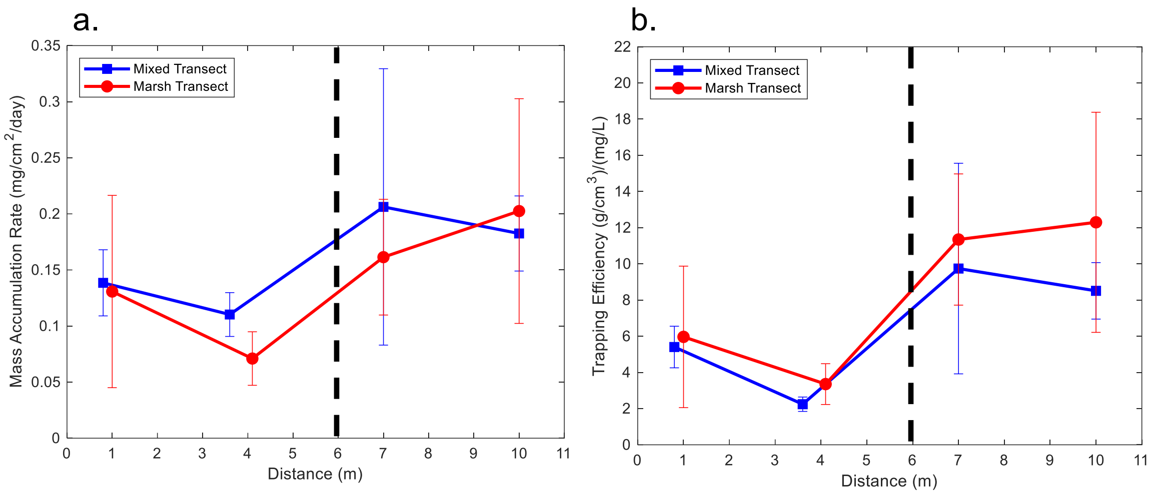

Figure 4. (a.) Mass accumulation rate (b.) and trapping efficiency with distance from the wetland edge for the mixed transect (blue) and the marsh-only transect (red). Error bars represent one standard deviation of replicate plots. The dashed line indicates the boundary between mangroves and marsh at the mixed transect.

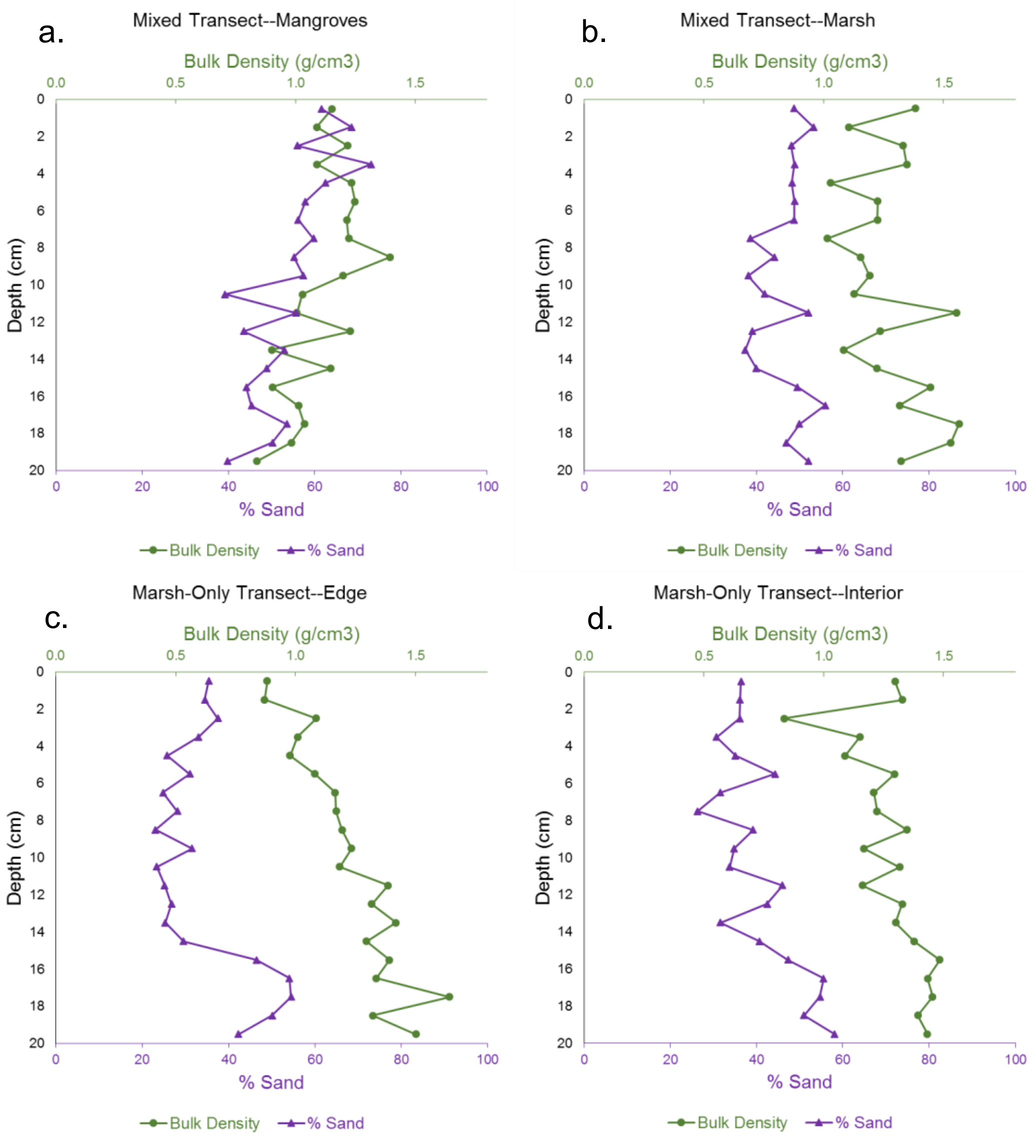

Figure 5: Bulk density and percent sand content for sediment cores at (a.) the fringing mangroves of the mixed transect, (b.) the adjacent marsh at the mixed transect, (c.) the edge site of the marsh-only transect, and (d.) the interior marsh of the marsh-only transect.

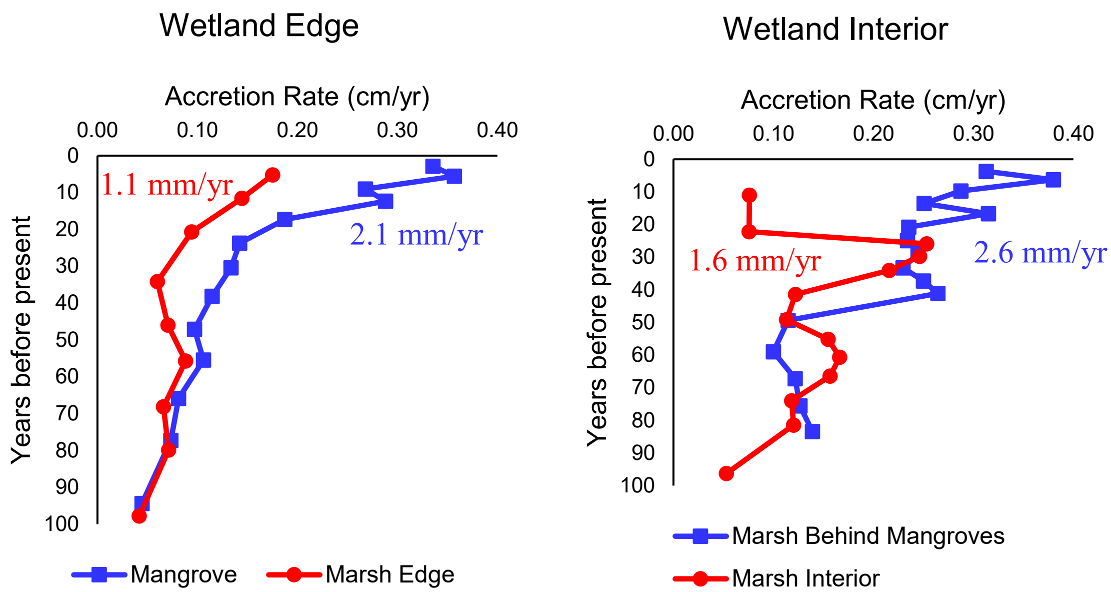

Figure 6: Accretion rate through time based on CRS Pb-210 radiochronology divided into (a.) the wetland edge cores and (b.) the wetland interior cores. Accretion rates accelerate through time for all sites except at the marsh interior site of the marsh-only transect. Labels indicate accretion rate averaged over the past 50 years in mm/yr.