

5-13-2022

The effects of short-term sea level rise on vegetation communities in coastal Mississippi

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The effects of short-term sea level rise on vegetation communities in coastal Mississippi

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A Thesis

Submitted to the Faculty of

Mississippi State University

in Partial Fulfillment of the Requirements

for the Degree of Master of Science

in Wildlife, Fisheries and Aquaculture

in the Department of Wildlife, Fisheries and Aquaculture

Mississippi State, Mississippi

May 2022

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2022

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Title of Study: The effects of short-term sea level rise on vegetation communities in coastal Mississippi

Pages in Study: 42

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Salt marshes are important habitats that provide many ecosystem services, but they are susceptible to the impacts of sea level rise (SLR), often resulting in emergent vegetation loss. In areas with enough sediment input, marshes can keep pace with SLR by gaining elevation or through upland migration. However, salt marshes in areas with limited sediment input, such as the Grand Bay National Estuarine Research Reserve, often cannot keep pace with sea level rise. Additionally, the rate of SLR is increasing making it more difficult for marshes to keep pace. To assess the short-term response of marsh vegetation to sea level rise, percent cover, stem density, and elevation, data from 2016 to 2020 in four different marsh elevation zones were analyzed in this study. Results demonstrated that the four marsh elevation zones are responding disparately to SLR. These findings indicate that it is imperative to implement restoration plans to account for site variability to conserve these vital habitats.

DEDICATION

To my grandmother, “G”,

Thank you for showing by example that faith and family should serve as the cornerstones of one’s life. I will always cherish the bond we shared in our love of books and learning and how you embraced my job, my thesis, and my desire to educate others. Thank you for all you were and all you have helped me become.

ACKNOWLEDGEMENTS

I would like to thank Dr. Eric Sparks for his patient guidance throughout this process and Dr. Ayesha Gray for the opportunity to work with the Grand Bay NERR during graduate school. Thanks to Dr. Jonathan Pitchford and Dr. Mark Woodrey for participating as committee members, lending advice and insight on this project at every step. Special thanks to Kimberly Cressman, NERR “R genius,” for providing invaluable help and guidance with the data processing code and statistical analyses. From proofreading several drafts and volunteering to help with my presentation, to being my “stay late buddy” on some of my longest nights—Sandra Bilbo, thank you for contributing so much to this project and being there for me.

To my Grand Bay NERR “family,” for their constant encouragement, sharing of their own graduate school experiences to make the bleak days better, and making fun field memories—I cannot thank you enough. To my “Tea Time” friends who never allowed me to go through hard times alone, giving me the necessary distractions when things got bumpy, and celebrating each accomplishment, big or small, you are all truly the best.

Lastly, to my biggest cheerleaders, my parents—your unconditional love and unwavering belief in me were constant sources of motivation throughout my graduate school journey. Despite all the hiccups, you were always there by my side to cheer me on. Mom and Dad, this would not have been possible without you—WE DID IT!

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CHAPTER I
THE EFFECTS OF SHORT-TERM SEA LEVEL RISE ON VEGETATION COMMUNITIES
IN COASTAL MISSISSIPPI

Introduction

When paired with coastal development, higher temperatures, increased storm intensity and frequency, and alterations in riverine and freshwater input, sea level rise (SLR) can be a significant stressor in reshaping coastal landscapes (Enwright et al., 2016; Gabler et al., 2017; Kirwan & Megonigal, 2013; Nicholls & Cazenave, 2010; Scavia et al., 2002). Sea level rise has been documented to be accelerating since 1901 (Church et al., 2013), and is expected to continue accelerating (IPCC, 2010). Whereas coastal marshlands survived previous periods of SLR (Redfield & Rubin, 1962), increases in sea level will likely result in approximately 20-50% global loss of marshland by the end of the century (Ganju et al., 2004; Ganju & Schoellhamer, 2006; Geyer et al., 2001; French, 2006). Increases in sea level alter frequency, depth, and duration of flooding in tidal wetlands (Church et al., 2013; Scavia et al., 2002), resulting in the stress and loss of valuable coastal habitats (Day et al., 2011; Nicholls et al., 1999; Nicholls et al., 2007; Turner & Boyer, 1997). It is particularly challenging to plan for SLR because impacts vary over small spatial and temporal scales (Christiansen et al., 2000; Temmerman et al., 2003).

The combination of rising water levels and sinking land (i.e., subsidence) results in what is often termed relative sea level (RSL) rise (Reed, 1995; Stevenson et al., 1986). In addition to changes in sea level, a major driver of the variation of relative sea level can often be attributed to

subsidence (Emery & Aubrey, 1991; Milliman & Haq, 1996; Rovere et al., 2016). Relative sea level rates are often 2 to 3 times higher than rates of SLR alone due to subsiding land (Church and White, 2011; Douglas, 2001; Holgate, 2007; Houston and Dean, 2011), which subsequently can have significantly greater impacts on coastal ecosystems (Kemp et al., 2015).

In temperate zones, salt marshes exist at the interface between land and marine waters, and although they can thrive and be highly productive in dynamic and physically stressful conditions, increases in inundation time and frequency can result in habitat conversion and/or loss of marsh function (Alongi, 2009; Ibanez et al., 2012; Kirwan et al., 2016; Twilley & Day, 2012). These ecosystems provide valuable ecosystem services such as shoreline protection (Gedan et al., 2009), water quality improvement (Sparks et al., 2015), and carbon sequestration (Bruno and Bertness, 2001; Chmura et al., 2003; McLeod et al., 2011). Salt marsh productivity depends on numerous factors (Mendelssohn & Morris, 2000), however, site elevation relative to mean sea level (MSL) is ultimately one of the most important (Kirwan and Guntenspergen, 2015; McKee and Patrick, 1988; Morris et al., 2002). Marsh plant productivity is species-specific, whereby each species has its own optimal inundation threshold, and when prolonged standing water levels surpass this threshold reductions in productivity and valuable ecosystem services can occur (Bertness, 1991; Chmura et al., 2003; Soulsby et al., 2013; Torfs et al., 1996). Therefore, an improved understanding leading to more cost-effective management of coastal marshes is critical to maintaining these important coastal ecosystems (Cahoon & Guntenspergen, 2010).

Salt marshes are characterized by prominent vegetative zonation patterns across elevation gradients and each respond to SLR disparately (Chapman, 1974). These zonation patterns occur because species will occupy areas within their optimal salinity and inundation thresholds

(Bertness, 1991; Bertness, 1992; Bockelmann et al., 2002; Jeom-Sook & Jong-Wook, 2018; Pennings & Callaway, 1992; Thompson, 2021). Along the northern Gulf of Mexico, the three types of marsh zones (high, mid, and low marsh) contain unique vegetative communities dependent upon species-specific physiological tolerances to salinity and inundation (Battaglia et al., 2012; Chapman, 1974; Visser et al., 2012). The most abundant marsh plants in the southeastern United States are *Juncus roemerianus* and *Spartina alterniflora*, with *Juncus roemerianus* occupying higher elevations and *Spartina alterniflora* dominating lower elevations and fringing marsh zones (Eleuterius, 1973, 1976; Pennings et al., 2005; Pomeroy et al., 1981; Stout, 1984; Touchette et al., 2009; Wiegert & Freeman, 1990).

The high marsh is a supratidal region along the transition from mid marsh to uplands, dominated by *Spartina patens* and/or *S. spartinae* with a variety of other species present and typically flooded during extreme tidal events and severe coastal storms (Eleuterius, 1973). Mid marsh is an irregularly inundated region comprised primarily of *J. roemerianus* with *S. alterniflora* in lower elevation areas such as along tidal creeks or along the seaward boundary of the marsh (Eleuterius 1973; Wiegert and Freeman, 1990). Low marsh is frequently flooded by daily tides and dominated by the less competitive, stress-tolerant *S. alterniflora* (Eleuterius 1973, Bertness, 1991; Bertness & Ellison, 1987).

The northern Gulf Coast is highly vulnerable to RSL due to a combination of physical and economic factors such as rapid urbanization and increasing potential impacts of projected SLR in the region (IPCC, 2007a). Small increases in sea level can contribute to vertical accretion rates (Cahoon, 2006) by allowing more sediment trapping on the marsh platform and stimulating plant growth with increased nutrient availability (Morris et al. 2002). Nonetheless, continued increases in SLR rate may surpass species inundation tolerances causing increased stress on

marsh vegetation, inhibiting its growth, and ultimately resulting in insufficient vertical accretion rates to maintain elevation (Morris, 2002; Nyman et al. 1993). In response to increasing SLR, marsh vegetation communities migrate upslope given a lack of barriers to this landward movement (Battaglia et al., 2012; Enwright et al., 2016). However, when these coastal habitats are caught between barriers (i.e., anthropogenic and natural) and rising seas, species occupying higher elevations may be subjected to coastal squeeze (Doody, 2004; Fagherazzi et al., 2019; French, 1997; Torio & Chmura, 2013) resulting in the loss of high marsh as the mid and low marsh migrate upslope (Black and Veatch, 2006; Defra, 2003; Doody, 2012; Field et al., 2016; Pethick, 1993; Pontee, 2013). Coastal land managers and researchers recognize the need to understand the consequences of these changes on a local scale, and thereby conduct research that informs local scale conservation planning to allow the co-existence of the natural environment and human activities in the coastal zone (Mimura, 2013).

Although marshes were able to respond when SLR was once a longer process (Church et al., 2013; Redfield & Rubin, 1962), with projected rapid rises in sea level (IPCC, 2010) associated changes in vegetation could be compressed in shorter timescales (NOAA, 2018) and marshes may be unable to keep pace (Crosby et al., 2016). Because the northern Gulf of Mexico has some of the highest rates of coastal erosion and wetland loss in the United States (Morton et al., 2004, 2005), SLR adaptation has been identified as a critical priority by the coastal conservation community (Cahoon, 2014). Specifically, conducting empirical field-based studies to understand the effects of SLR across this region has been noted as a high priority (Bakker et al., 1993; Burkett, 2008; Duke et al., 2007; Fitzgerald et al., 2008; Hall, 2011).

To address this research priority, we collected and analyzed changes in marsh vegetation from 2016 to 2020 across four marsh habitats (i.e., salt panne, high-elevation *Juncus*

roemerianus, low-elevation *Juncus roemerianus*, and *Spartina alterniflora*) along an elevation gradient within the Grand Bay National Estuarine Research Reserve (GNDNERR). Based on my review of the literature and my knowledge of the Grand Bay NERR, we developed a null-hypothesis stating that there will be no changes in marsh surface elevation over time. Alternatively, there will be changes in marsh surface elevation over time. The second null-hypothesis was there will be no changes in vegetation community over time. Alternatively, we hypothesized there will be changes in vegetation community over time. The main objective of this study was to determine the effects of short-term sea level rise on vegetation community structure in four marsh sites within coastal Mississippi.

Methods

Study site

This study occurred within the Grand Bay National Estuarine Research Reserve (GNDNERR). The GNDNERR is a marine-dominated estuary located in southeastern Mississippi, encompassing approximately 7,400 ha (18,000 acres) of protected coastal, wetland, and terrestrial habitats (Alizad et al., 2016; Gilmer et al., 2011; Pitchford et al., 2022). This area is dominated largely (>90%) by *Juncus roemerianus*, however, other dominant salt marsh species are present such as *Spartina patens* and *Spartina alterniflora*, with the latter occupying the fringe of marshes (Archer et al., 2021; NOAA, 2016; Wu et al., 2017).

The GNDNERR historically received freshwater inputs from the Escatawpa River, but it naturally redirected to the Pascagoula River, leaving the region a retrograding delta with no sediment input from a major riverine system (Eleuterius and Criss, 1991; Otvos, 1985; Wu et al., 2017). Today, pine savannas and brackish marshes dominate the north while more saline waters from Mississippi Sound dominate the south (Thompson, 2021). The diurnal tidal range in the

area averages 0.6 meters while the annual tidal range is approximately 1.1 meters (Dillon & Walters, 2007; Eleuterius and Criss, 1991; Otvos, 2007a).

The closest National Water Level Observation Network tide gauge station to the GNDNERR is located in Dauphin Island, Alabama (NOAA, 2018). Since 1966, SLR has increased by 279mm and the speed of SLR has accelerated over the last decade (NOAA, 2018). Therefore, increases in sea level was once a process that took approximately 22 years to rise by 152mm, but this phenomenon is currently rising by over 25mm every 2 years (NOAA, 2018). The current rate of SLR in the northern Gulf of Mexico is 3.74mm/yr (NOAA, 2022), and projections estimate sea levels will rise by another 152mm in the next fifteen years at Dauphin Island, Alabama (Pitchford et al., 2022; USACE, 2021).

To better understand the impacts of SLR and increased inundation of coastal habitats the GNDNERR established a Sentinel Site Program in 2010. The program evolved from the System Wide Monitoring Program (SWMP) and focuses on changes in sea level, salt marsh elevation and vegetation change, shoreline erosion, water level, surface water quality, and meteorological data monitoring (Office of Ocean and Coastal Resources, 2012). Five sites were selected along a north-south transect spanning 3.5 miles from a freshwater marsh at the north end to a low elevation marsh at the south end. However, I focused my analyses on data collected from the four lowest elevation sites that encompassed the following four habitat types: salt panne, *Juncus roemerianus*, low elevation *Juncus roemerianus*, and *Spartina alterniflora* (Figure 1.1, 1.2). Each was named by abbreviating the genus and species names of the most abundant species found at a site (i.e., JURO stands for *Juncus roemerianus*, SPALT stands for *Spartina alterniflora*) (USDA, 2007). JURO High was placed along the upland to salt marsh ecotone and includes some salt panne habitat, while JURO Mid is mainly a *J. roemerianus* marsh with some

interspersed salt panne habitats (Figure 1.2a). JURO Low is dominated by *J. roemerianus* but has several *S. alterniflora* meadows present (Figure 1.2b). Most of this site's eastern shoreline was characterized by an erosional edge and fringing *S. alterniflora*. Being the lowest in elevation, SPALT was most regularly inundated and dominated by *S. alterniflora* with small patches of *J. roemerianus* (Figure 1.2c).

At each of the four sites, three transects were established to understand long-term change in vegetation community composition (Lynch et al., 2015; Office of Ocean and Coastal Resources, 2012). Furthermore, Surface Elevation Tables (SETs) were established to enable precise measurements of the marsh surface to gain a better understanding of marsh elevation change (e.g., sediment accretion and subsidence). Each transect was adjacent to each SET, with seven to fourteen plots per transect totaling 123 plots along 12 transects (Figure 1.2a, b, c). Data collected from these transects were used in this study.

Field Measurements

Each plot along transects was marked with four corner PVC poles to delineate a permanent 1 m² monitoring area that was used to monitor vegetation community changes over time. Annual vegetation and elevation monitoring occurred at these sites between July and August from 2015 to 2020. During each sampling event, the elevation of each plot corner was collected using a Trimble R10 real time kinematic (RTK) GPS (Sunnyvale, CA), which had a horizontal precision of 20 mm and a vertical precision of 30 mm.

Within each 1 m² plot, vegetation percent cover, stem density, and average canopy height were collected. Percent cover was determined by visually estimating the proportion of ground covered by each vegetation species within each plot (Moore, 2009). As with any field data, visual percent cover estimates are subject to human error (Martin et al., 2021). To account for

this error, the procedure established by Brower et al., 1990 was used to standardize estimations for percent cover data collection, having the same staff members collect these data for each sampling year. Stem density was determined by counting the number of stems for each vegetation species within boundaries of a 0.0625 m² subsection of a quadrat (Figure 1.3).

Analysis

Vegetation Statistical Analyses

The Quality Assurance Quality Control (QAQC) procedures for vegetation data established by the NERR Biomonitoring Workgroup (2020) were implemented prior to analyses; blank spaces were converted to zeros. Due to staff changes and potential variable field methodology before 2016, only data from 2016-2020 were analyzed in this study. A Shapiro-Wilk test was used to evaluate the underlying distributions of the elevation data and percent cover and stem density of the two most abundant species per site (*J. roemerianus* and *S. patens* for higher elevation sites and *J. roemerianus* and *S. alterniflora* for lower elevation sites) for each sampled year. Square root and fourth root transformations were attempted on non-normally distributed data, with each attempt tested for normality using the Shapiro-Wilk test. Despite transformations, percent cover, stem density, and elevation data did not approach normality. Prior to running further analyses, each species' percent cover and stem density data were averaged by transect (i.e., all points along each transect were averaged together) for each sample year.

Linear regressions were used to quantify the relationship between sampling year and percent cover and stem density of the dominant species. Plots of residuals were examined to determine whether they met the normality and homogeneity of variances assumptions of this approach. When the plots did not indicate substantial deviation from normality, the results of the

linear regression were retained. This was the case for *J. roemerianus* percent cover at JURO Mid, JURO Low, and SPALT. When residuals did violate normality, the non-parametric Kendall's Tau was used instead. This was the case for all species' stem density at JURO High, JURO Mid, JURO Low, and SPALT, all *S. patens* and *S. alterniflora* percent cover, and *J. roemerianus*' percent cover at JURO High.

To examine temporal patterns in the vegetation community, within each study site over the five-year sampling period, Bray-Curtis dissimilarity matrices were calculated using the data that were averaged by transect. Analysis of similarity (ANOSIM) was utilized to test for significant differences in vegetation community by transect at each site. ANOSIM was also used to test for differences in the vegetation community between the first and last year of data collection. Similarity of percentages (SIMPER) was used to compare the Bray-Curtis dissimilarity matrices of the first and last year's percent cover and stem density data to identify which species contributed most to significant differences indicated by ANOSIM. Nonmetric multidimensional scaling (nMDS) was used to visualize Bray-Curtis dissimilarity matrices for all sites over time; stress values from nMDS measured the degree to which the distance between samples in reduced dimensional space (i.e., 2-dimensions) corresponded with the actual distance between samples (Letten, 2016). PRIMER's guidance on stress values indicates that any value ≥ 0.3 is arbitrary, ≥ 0.2 is suspect, ≤ 0.1 is fair, and ≤ 0.05 is a good fit (Clarke & Gorley, 2015). Visualizations were made with the ggplot2 package (Wickham, 2016) and analyses were performed with the vegan package (Oksanen et al., 2020) using R 4.1.1 (R Core Team, 2021) in RStudio (RStudio Team, 2020). *A priori* significance for all tests was set at $\alpha < 0.05$.

Results

Over the five years of the study, a total of 36 plant species were found across all sampling plots. JURO Mid had the highest plant diversity with 30 species. The next most diverse site was JURO High with 26 species, followed by JURO Low (3 species) and SPALT (2 species; Table 1.1). There were no significant differences in percent cover, stem density, and diversity within each site ($p > 0.12$; Table 1.2). Therefore, only comparisons among years and between sites for percent cover, stem density, and elevation data are described below.

Elevation

North American Vertical Datum of 1988 (NAVD88) elevation ranged from 0.049m to 0.276m at the lowest site (e.g., SPALT), -0.017m to 0.776m at mid sites (e.g., JURO Mid and JURO Low), and 0.224m to 0.763m at high sites (JURO High). The four sites split into two distinct groups: JURO High and JURO Mid were higher elevation sites while JURO Low and SPALT are lower on the elevation gradient (Figure 1.4).

Percent Cover

Juncus roemerianus had the highest percent cover across all sites (Table 1.1). The results of this study indicated there were no significant trend in percent cover at JURO High and JURO Mid from 2016 to 2020 (Table 1.4), however, percent cover of *J. roemerianus* significantly declined at both JURO Low and SPALT sites across the five years of this study. *Spartina patens* percent cover declined significantly across the years at both JURO High and JURO Mid (Table 1.4). *Spartina alterniflora* was only found at the two sites that were lowest in elevation, JURO Low and SPALT, but showed no significant change in percent cover over the course of this study (Tables 1.1, 1.4; Figure 1.5b).

Stem Density

Stem density showed no change over time for either *J. roemerianus* or *S. alterniflora* at any site ($p > 0.05$, Table 4, Figure 1.6 panels a, b). However, *S. patens* stem density showed a significant decrease over time at the JURO Mid site (i.e., $p = 0.008$, Table 1.4, Figure 1.6c).

ANOSIM & SIMPER

The ANOSIM analysis indicated a significant shift in the plant community (defined here as a combination of species-specific percent cover and stem density data) at JURO Low and SPALT from 2016 as compared to 2020 ($p \leq 0.01$, Table 1.3, 1.4). Results from the SIMPER analysis indicated *J. roemerianus* percent cover as the driver of these significant shifts over time at JURO Low and SPALT (Figure 1.5a, Tables 1.3, 1.4). No significant changes in the plant communities were detected at the JURO High and JURO Mid sites ($p > 0.12$, Table 1.3) from 2016 as compared to 2020. Although *S. patens* percent cover significantly changed at JURO High and JURO Mid, no significant changes in the plant community at these sites were detected by ANOSIM.

Nonmetric multidimensional scaling

The nonmetric multidimensional scaling (nMDS) analysis displayed many notable differences in the overall species compositional patterns at the sites over time (Table 1.3, Figures 1.7, 1.8). JURO High and JURO Mid points were grouped together, indicating these sites were similar to one another. The points from these two sites were also more spread out than the points from JURO Low and SPALT, indicating more community variability at the higher sites than the lower ones (Figures 1.7, 1.8). Conversely, JURO Low and SPALT points were more separated

from each other, indicating the two sites were less similar to one another than JURO High to JURO Mid (Figures 1.7, 1.8).

In these analyses, Axis MDS1 seemed to represent a site/elevation gradient (Figure 1.8) with sites separating by species: *S. patens* was associated with higher sites (i.e., plotted along the left side of MDS1), while *S. alterniflora* was associated with lower sites (i.e., plotted along the right side of MDS1). Axis MDS2 seemed to represent differences between years (Figure 1.7), with 2017, 2019, and 2020 points being placed between 2016 and 2018 points, which were placed at the top and bottom of the nMDS graph, respectively. The nMDS stress value was 0.139, indicative of a fair fit (Figures 1.7, 1.8; Clarke and Gorley, 2015). Resulting differences in vegetation metrics (i.e., percent cover, stem density, and elevation) among sites are discussed below.

Discussion

There are a wealth of studies focused on understanding vegetation dynamics within the GNDNERR that suggest marsh resilience in this area heavily depends on the magnitude of SLR over the next several decades (Archer et al. 2021; Raposa et al. 2016; Wu et al. 2017; Wu et al. 2020). Physiological stressors (e.g., increases in inundation, salinity, etc.) can cause functional loss (Ellison et al., 2005), and, therefore alter the presence and movement of vegetative communities over time (Baustian et al., 2012; Archer et al., 2021). Considering SLR was once a longer-term process that took approximately 22 years to rise by 152mm, this phenomenon is rising by over 25mm every 2 years (NOAA, 2018) in Dauphin Island, Alabama. In 2018, NOAA data from the area showed a relative sea level trend of 3.74 ± 0.58 mm/year; however, current NOAA data shows a relative sea level trend of 4.13 ± 0.59 mm/year in the area (NOAA, 2022). Given that most previous studies have focused on long-term projections, this shorter-term study

filled an informational need. Therefore, completion of this work can improve our understanding of the coastal processes taking place at the GNDNERR. Ultimately, this will assist in the development of effective management and mitigation strategies to restore ecological functions and enhance resilience to SLR.

Results of my study indicated that overall vegetation communities, percent cover, and stem density are changing at GNDNERR, and sites are responding disparately to SLR over short timeframes. Over a relatively short timescale of 5 years, overall plant communities in higher elevation sites (JURO High and JURO Mid) showed no significant changes, but lower elevation sites (JURO Low and SPALT) significantly changed over time (Table 1.3). The sites clustered together into two distinct groups: JURO High and JURO Mid; JURO Low and SPALT (Figures 1.7, 1.8). This grouping indicated that sites higher in elevation are more likely to be similar to each other while the lower sites in elevation are likely to be more similar to themselves.

Noticeable changes in percent cover and stem density of dominant species were also observed over a relatively short timescale of 5 years, and the magnitude of these differences varied by site and associated biophysical characteristics. Since *J. roemerianus* had the largest elevation span of the marsh plants in this study (i.e., -0.286 to +1.2m NAVD88) (Constantin et al., 2019), its presence at all sites was expected. However, the documented negative trends in percent cover at the lowest elevation sites (JURO Low and SPALT) over a short time period was not anticipated (Tables 1.3, 1.5). Declines in *J. roemerianus* can occur when water levels exceed the species' inundation tolerance 25% of the time (Voss et al. 2013).

The precision of elevation measurements (RTK) was precise, but generally not precise enough (i.e., error = 1-3cm) to capture elevation change over the five-year period, unless there were large erosion or sedimentation events. Considering the annual rates of change and the

precision error of the RTK (i.e., 1-3 cm), the ability of the RTK to detect a meaningful change in elevation over the short time period of five years is limited. Therefore, there were no detectable changes in elevation at any of the sites over the duration of this study. Using the SETs, which are more precise, Pitchford et al. (2022) found the annual rate of elevation change from 2012-2016 was 2-5 mm/yr for all the sentinel sites, with JURO High being the only site with a significantly lower elevation change rate (i.e., 0.54 mm/yr) compared to the SLR rate (3.5mm/yr), and JURO Low having a significantly higher elevation change rate (5.45 mm/yr) compared to SLR rate. Therefore, plants found at JURO high are experiencing inundation at increasingly higher levels due to the elevation change rate being less than SLR rates in the area (Pitchford et al. 2022), which could lead to elevated stress on plants in this zone and could explain the declines in *S. patens* percent cover and stem density observed in my study. Further, the higher elevation change rate at JURO Low, as determined by Pitchford et al. (2022), could explain why *J. roemerianus* was still present at the site in 2020, despite significant declines in *J. roemerianus* percent cover over time.

Research completed in New England National Estuarine Research Reserves found that less flooded areas supported more high marsh species and species richness increases from the low marsh to upland edge (Burdick et al., 2020). Along the 3.5 mile transect in my study, the lower elevated sites (JURO Low and SPALT) likely faced increased physiological stress (i.e., inundation and salinity stress from daily tides) that, surpass high marsh species tolerances, making it difficult for these areas to support more diverse species. This could explain declines in species diversity and richness at JURO Low and SPALT (i.e., comprised of 1-3 species) while species richness and diversity was higher at JURO High and JURO Mid (i.e., comprised of 25-30 species) during the study period.

Moreover, upland ecotones with NADV88 elevations between 0.286 and 0.542 m are optimal for *S. patens* growth; however, productivity would decline if the inundation frequency in these areas increased (Archer et al., 2021; Walkup, 1991). *S. patens* percent cover declined at the two highest elevation sites (JURO Mid and JURO High), while *S. patens* stem density also declined at the JURO Mid site. The declines in percent cover, stem density, or both documented in this study could likely be attributed to both plants (i.e., *J. roemerianus* and *S. patens*) experiencing inundation above their ideal limits (Morris 2006; Warren & Niering, 1993; Wu et al., 2017). This is especially true at JURO High and JURO Mid, where elevation change rates (Pitchford et al., 2022) may not be keeping pace with SLR and associated physiological stressors (Alongi, 2009; Ellison et al., 2005; Ibáñez et al., 2012; Kirwan et al., 2016; Twilley & Day, 2012).

In contrast, there were no significant changes in percent cover or stem density in the more inundation-tolerant *S. alterniflora*. With optimal elevation thresholds between 0.084 and 0.31m NADV88 (Archer et al., 2021), *S. alterniflora* can tolerate high tidal amplitudes, however, if there are prolonged standing water levels above its optimal threshold, reductions in *S. alterniflora* productivity can occur (Bertness, 1991). Wu et al. (2017) found that coastal wetlands may be resilient to SLR by 2050 under intermediate-high SLR scenarios, but it is very likely that highly vulnerable coastal wetlands like GNDNERR could collapse by the end of the century (Alizad et al., 2018). Therefore, continued monitoring of plant communities across the elevation gradient, including the percent cover and stem density of each species, is necessary to help understand the trajectory of the system.

Other factors may have also played a role in vegetation changes from 2016-2020. For example, in March 2018, a prescribed fire occurred at JURO High and could potentially explain

the negative trends seen in *S. patens* percent cover for that year of sampling. Also, interannual climate variability (Moreira et al., 2021) including variation in temperature and/or precipitation (Leclercq & Oerlemans, 2012; Malone et al., 2019; Oerlemans, 2005) may have occurred. Specifically, in September 2018, Tropical Storm Gordon made landfall just west of the Mississippi-Alabama border, where maximum water levels reached 2.1 feet above the MHHW (Brown et al., 2019). Therefore, during such major storm events, inundation tolerances are likely surpassed, and die-off can result. These results corroborate findings from research in New England National Estuarine Research Reserves, where typical high marsh species were becoming less abundant in high marsh regions as sea levels increased (Burdick et al., 2020).

Management Implications

Prescribed fire along the upland ecotone could encourage marsh migration by removing potential forest barriers and enhance marsh productivity (Constantin, 2015; Hacker, 2018). A previous study in McFaddin National Wildlife Refuge concluded that prescribed burning could increase rates of elevation gain while also slowing the rate of subsidence compared to a non-burned marsh (Cahoon et al., 2004). Since the GNDNERR is a retrograding delta without sediment input from an upstream riverine system (Eleuterius and Criss, 1991; Otvos, 1985), another restoration option coastal land managers should also consider is thin layer placement. This process enhances marsh resiliency to SLR by increasing tidal marsh elevation in areas with limited accretion (Raposa, 2020). A study in Narragansett Bay National Estuarine Research Reserve showed that after just a few months, thin layer placement (i.e., 7cm sediment layers) allowed vegetation cover to rapidly recover in both high and low marsh (Raposa, 2020). If effective mitigation plans are not implemented to facilitate adaptation for future rises in sea level, GNDNERR will likely have continued changes in vegetation community structure.

Future Research

Findings of this study demonstrate that each site is responding disparately to SLR. Therefore, it is imperative to implement restoration plans that can account for each site's variability and conserve these vital habitats. Despite contributing to the overall goal of the NERRS Sentinel Site Program, discerning the contributions of subsidence, plant-facilitated sediment accretion, and the relative sea level in the area were outside the focus of this study. Therefore, more studies are necessary to include these metrics and potentially aid in the development of effective mitigation plans that facilitate adaptation for future rises in sea level at the GNDNERR. Since vegetation monitoring is ongoing, data will be available for future analyses, perhaps to address these limitations and help determine the best mitigation solutions for long-term conservation of GNDNERR marsh. Inclusion of other metrics not included in this study, but measured as part of the Sentinel Site Program, such as canopy height, Surface Elevation Table (SET), marker horizon, and water quality data should be used to help interpret vegetation changes. Other percent cover methods (i.e., point-intercept method) can be used to develop tools (e.g., resilience assessments) to offer further insight into marsh vegetation resiliency. Additionally, without a protocol for visual percent cover in early sampling years, estimates were subject to human error, and measurements could have included both standing dead and living biomass. Nevertheless, with current methodology in place, monitoring must continue to better understand both long- and short-term changes in vegetation communities in response to SLR. The effect of acute events (e.g., hurricanes and wildland fires) reinforces the need to continue monitoring vegetation in GNDNERR. Considering these types of events are regular occurrences, a longer period of record for metrics like percent cover and community composition is needed to more clearly link marsh vegetation change to SLR.

Tables

Table 1.1 Percent cover measurements of all species found within the four study sites at the Grand Bay National Estuarine Research Reserve from 2016 to 2020. Dashes within cells represent species not found at sites while zeroes represent species typically identified at sites, but not present for that sample year.

Species	JURO High					JURO Mid					JURO Low					SPALT				
	2016	2017	2018	2019	2020	2016	2017	2018	2019	2020	2016	2017	2018	2019	2020	2016	2017	2018	2019	2020
<i>Juncus roemerianus</i>	43	28	16	17	18	56	14	18	20	28	62	32	17	26	32	32	29	17	19	18
<i>Spartina patens</i>	8	5	2	4	5	4	3	1	1	1	-	-	-	-	-	-	-	-	-	-
<i>Spartina alterniflora</i>	-	-	-	-	-	-	-	-	-	-	6	4	3	5	6	22	24	14	15	21
<i>Salicornia depressa</i>	0.6	0.4	0.4	0.4	0.4	0.1	0.1	0.3	0.3	0.4	-	-	-	-	-	-	-	-	-	
<i>Panicum virgatum</i>	3	3	1	1	1	0.1	0.1	0.1	0.1	0.3	-	-	-	-	-	-	-	-	-	
<i>Fimbristylis spp.</i>	1	0	0	0	0	0	0.1	1	2	1	-	-	-	-	-	-	-	-	-	
<i>Borrichia frutescens</i>	1	1	1	1	1	4	3	2	3	3	-	-	-	-	-	-	-	-	-	
<i>Distichlis spicata</i>	0.4	0.4	0	0.5	0.4	2	2	2	1	1	0.1	0.1	0.1	0.2	0.5	-	-	-	-	
<i>Cladium jamaicense</i>	3	1	1	1	1	4	3	1	2	1	-	-	-	-	-	-	-	-	-	
<i>Spartina spartinae</i>	1	3	1	1	2	1	1	1	1	2	-	-	-	-	-	-	-	-	-	
<i>Limonium carolinianum</i>	1	0.5	0.5	1	1	1	1	1	1	1	-	-	-	-	-	-	-	-	-	
<i>Digitalis spp.</i>	0.2	0	0.2	0	0	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
<i>Iva frutescens</i>	0.2	0.1	0	0	0	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
<i>Setaria parviflora</i>	0.5	0.1	0	0.5	0	0.1	0	0.1	0	0.1	-	-	-	-	-	-	-	-	-	
<i>Ipomoea sagittata</i>	0.1	0	0.2	0.4	0.1	0.1	0	0	0.1	0	-	-	-	-	-	-	-	-	-	
<i>Baccharis halimifolia</i>	0.2	1	0	0.1	0.2	1	0.1	0.3	0.7	0.1	-	-	-	-	-	-	-	-	-	
<i>Agalinus maritima</i>	0	0.1	0	0	0.6	0.1	0.2	0.3	0.4	0.4	-	-	-	-	-	-	-	-	-	
<i>Dicanthelium spp</i>	0	0.3	0.6	0	0	0	0.1	0.1	0.1	0.1	-	-	-	-	-	-	-	-	-	
<i>Solidago spp.</i>	0	0.1	0	0.1	1	0	0.1	0.1	0	0	-	-	-	-	-	-	-	-	-	
<i>Unknown</i>	0	0.3	0.5	0.4	0	0.1	0.2	0.3	0.3	0.1	-	-	-	-	-	-	-	-	-	

Table 1.1 (continued)

<i>Sabatia stellaris</i>	0	0	0.7	0.6	0	0	0.1	0.1	0.1	0	-	-	-	-	-	-	-	-	-
<i>Unknown 2</i>	0	0	0.1	0	0	-	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>Pinus elliottii</i>	0	0	0.1	0	0	0.1	0.1	0	0	0	-	-	-	-	-	-	-	-	-
<i>Batis maritima</i>	0	0	0	0.1	0	0	0	0	0	0.2	-	-	-	-	-	-	-	-	-
<i>Andropogon spp</i>	0	0	0	0.2	0.2	0	0	0	0	0.2	-	-	-	-	-	-	-	-	-
<i>Seutera angustifolia</i>	0	0	0	0	0.5	0	0	0	0	0.1	-	-	-	-	-	-	-	-	-
<i>Schoenoplectus spp</i>	0.1	0	0	0	0	-	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>Setaria parviflora</i>	-	-	-	-	-	0.1	0	0.1	0	0.1	-	-	-	-	-	-	-	-	-
<i>Ipomoea spp.</i>	-	-	-	-	-	0	0.1	0.3	0	0.1	-	-	-	-	-	-	-	-	-
<i>Phyla nodiflora</i>	-	-	-	-	-	0.4	0.1	0.3	0	0.1	-	-	-	-	-	-	-	-	-
<i>Aster spp.</i>	-	-	-	-	-	0.2	0	0	0	0	-	-	-	-	-	-	-	-	-
<i>Sagittaria latifolia</i>	-	-	-	-	-	0	0.1	0	0	0	-	-	-	-	-	-	-	-	-
<i>Ilex spp.</i>	-	-	-	-	-	0	0.1	0.1	0.3	0	-	-	-	-	-	-	-	-	-
<i>Rhexia spp</i>	-	-	-	-	-	0	0	0.1	0.1	0	-	-	-	-	-	-	-	-	-
<i>Ilex vomitoria</i>	-	-	-	-	-	0	0	0	0	0.6	-	-	-	-	-	-	-	-	-
<i>Paspalum spp.</i>	-	-	-	-	-	0	0	0.1	0	0	-	-	-	-	-	-	-	-	-

Table 1.2 Analysis of similarities (ANOSIM) summary table of year on Bray-Curtis dissimilarity matrices to test for significant differences between each site's three transects

Site	P-value
JURO HIGH	0.121
JURO MID	0.22
JURO LOW	0.99
SPALT	0.31

Significant p-values ($p < 0.05$) are noted with an asterisk.

Table 1.3 Analysis of Similarities (ANOSIM) summary table from 2016-2020 data collection at individual sites

Site	P-value
JURO HIGH	0.2201
JURO MID	0.1201
JURO LOW	0.0001*
SPALT	0.0101*

Significant p-values ($p < 0.05$) are noted with an asterisk.

Table 1.4 Results of linear regression models (LM) and non-parametric Kendall's Tau (KT) for dominant vegetation species over time.

SITE	SPECIES	VARIABLE	TEST	MULTIPLE R-SQUARED	P-VALUE	SLOPE
JURO HIGH	<i>J. roemerianus</i>	Percent cover	KT	-	0.7287	0.2679
	<i>S. patens</i>	Percent cover	KT	-	0.0465*	-0.3571
	<i>J. roemerianus</i>	Stem density	KT	-	0.3211	0.3846
	<i>S. patens</i>	Stem density	KT	-	0.8037	-0.0714
	-	Elevation	KT	-	0.2763	-0.0017
JURO MID	<i>J. roemerianus</i>	Percent cover	Lm	0.1655	0.0739	0.0333
	<i>S. patens</i>	Percent cover	KT	-	0.0064*	-0.2196
	<i>J. roemerianus</i>	Stem density	KT	-	0.0600	0.4924
	<i>S. patens</i>	Stem density	KT	-	0.0076*	-0.0833
	-	Elevation	KT	-	0.1966	0.0032

Table 1.4 (continued)

JURO LOW	<i>J. roemerianus</i>	Percent cover	Lm		0.3033	0.0195*	-0.2315
	<i>S. alterniflora</i>	Percent cover	KT		-	0.0533	-0.2083
	<i>J. roemerianus</i>	Stem density	KT		-	0.8043	-0.0619
	<i>S. alterniflora</i>	Stem density	KT		-	0.6207	0.0139
	-	Elevation	KT		-	0.0748	-0.0028
SPALT	<i>J. roemerianus</i>	Percent cover	Lm	0.4174	0.0055*	-0.8888	
	<i>S. alterniflora</i>	Percent cover	KT	-	0.6915	-0.1543	
	<i>J. roemerianus</i>	Stem density	KT	-	0.7287	0.1026	
	<i>S. alterniflora</i>	Stem density	KT	-	0.6915	0.0476	
	-	Elevation	KT	-	0.1133	-0.0030	

Regression results for each Sentinel Site, averaged by transect, over five year sample period. Adjusted R² and p-values are shown for linear models. Significant p-values (p < 0.05) are noted with an asterisk.

Table 1.5 Results from Similarity of Percentages (SIMPER) showing the vegetation species contributing to significant changes in species cover and stem density between first and recent year at each site

SITE	SPECIES	CUMULATIVE CONTRIBUTION (%)	AVERAGE SITE DISSIMILARITY (%)
JURO HIGH	<i>J. roemerianus</i> pc*	43.13	17.64
	<i>S. patens</i> pc*	70.02	2.02
	<i>J. roemerianus</i> sd*	65.08	2.70
	<i>S. patens</i> sd*	58.47	6.27
JURO MID	<i>J. roemerianus</i> pc*	50.08	24.92
	<i>J. roemerianus</i> sd*	62.86	6.36
	<i>C. jamaicense</i> pc	68.22	2.66
	<i>S. patens</i> pc*	71.87	1.82
JURO LOW	<i>J. roemerianus</i> pc*	79.34	21.69
SPALT	<i>J. roemerianus</i> pc*	77.88	12.29
	<i>J. roemerianus</i> sd*	57.77	4.28

The above species and covers together explain over 80% of the variation between first year and most recent year of each site. Species that significantly differ between years ($p < 0.05$) are noted with an asterisk.

Figures



Figure 1.1 Location of Grand Bay National Estuarine Research Reserve (GNDNERR) and the four study sites along a 3.5 mile north to south transect: JURO High, JURO Mid, JURO Low, and SPAL

The inset map shows the boundary of the Reserve and its geographic location within the north-central Gulf of Mexico.

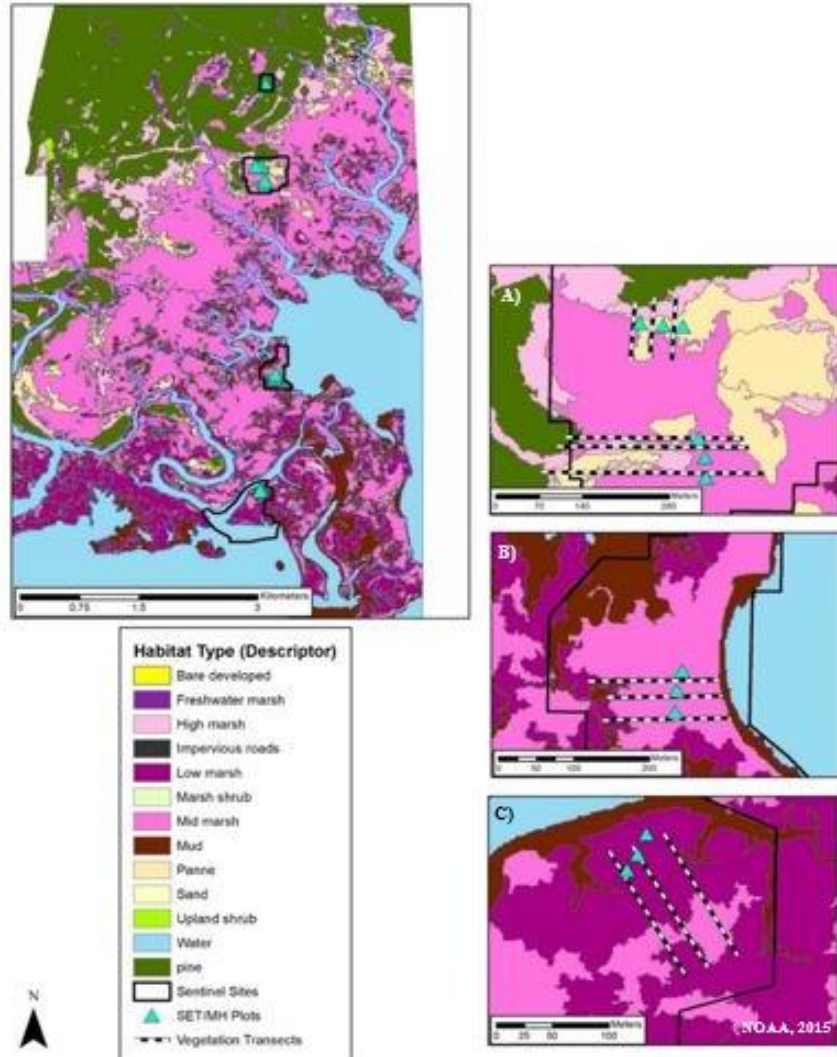


Figure 1.2 Overview and expanded habitat map of Grand Bay National Estuarine Research Reserve (GNDNERR) and study sites along a 3.5 mile transect

Habitat map showing the location of the four sites (panels A, B, C) and their habitats along the 3.5 mile transect. Panel A portrays JURO High and JURO Mid, with mid marsh and salt panne species. Panel B portrays JURO Low, dominated by mid marsh vegetation and patches of low marsh species. Panel C displays SPALT, dominated by low marsh vegetation and patches of mid marsh species.

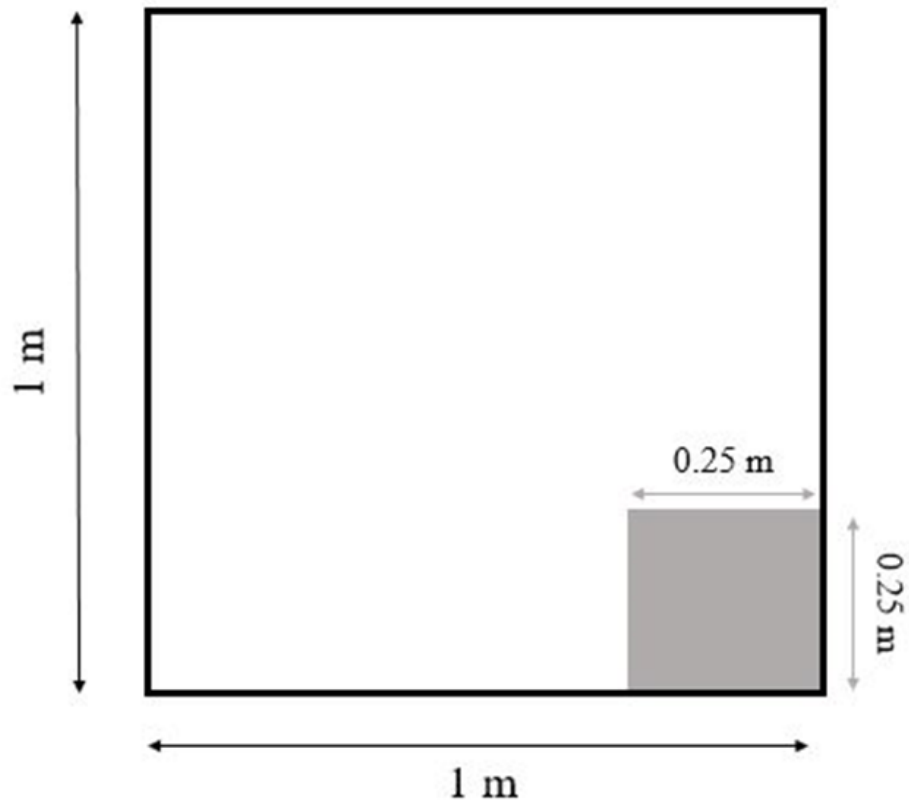


Figure 1.3 Diagram of a 1 m² quadrat marked with 0.0625 m² subsection (i.e., grey box) used for stem density data collection

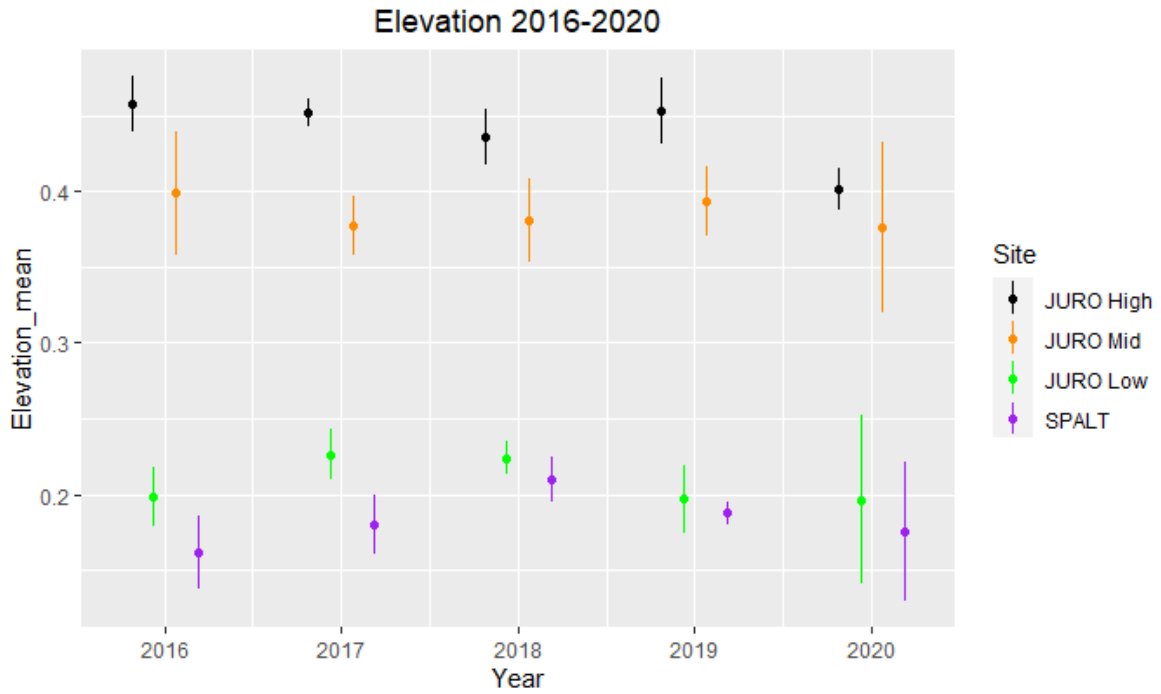


Figure 1.4 Elevation changes from 2016 to 2020 at all of the study sites (datum NAVD88)

Each point represents the mean of the three transects and the error bars represent mean \pm 1 standard deviation. Sites are color coded whereby JURO High is black, JURO Mid is dark orange, JURO Low is green, and SPALT is purple.

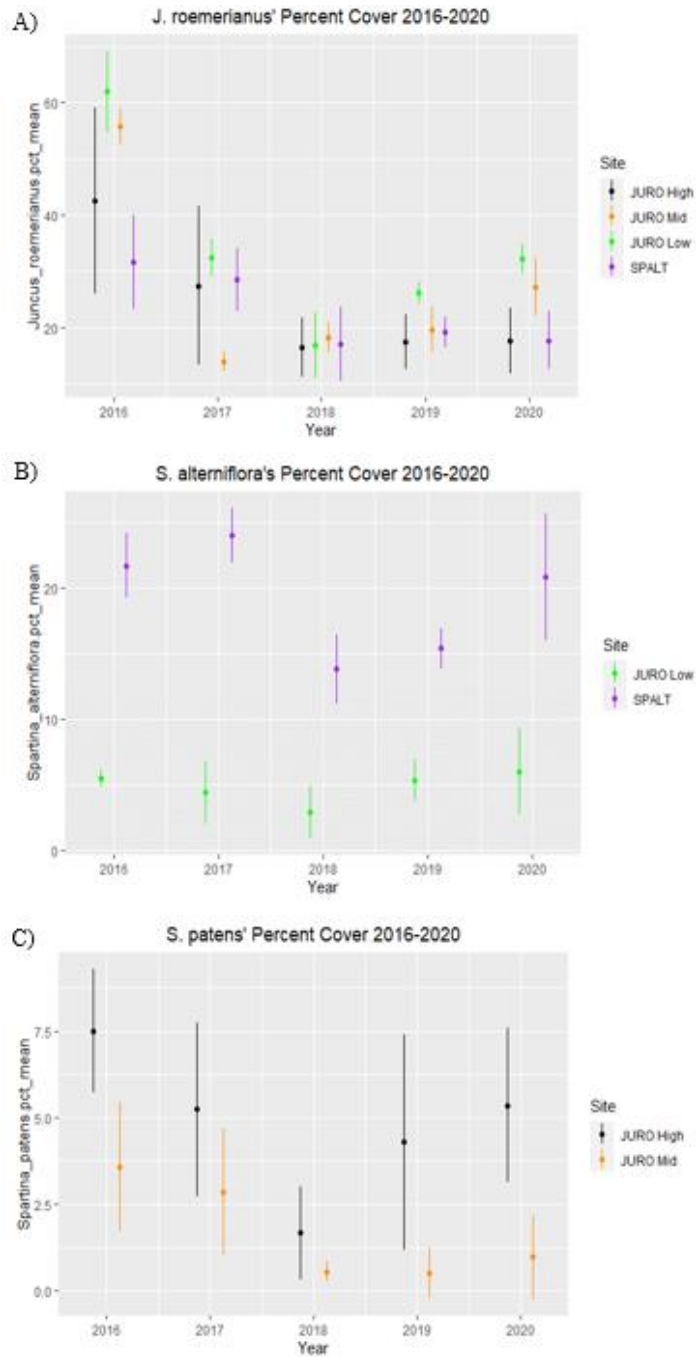


Figure 1.5 Percent cover changes for *J. roemerianus*, *S. alterniflora*, and *S. patens* from 2016 to 2020 at each study site (panels A, B, C, respectively)

Each point represents the mean of the three transects and the error bars represent mean \pm standard deviation. Each site is color coded whereby JURO High is black, JURO Mid is dark orange, JURO Low is green, and SPALT is purple.

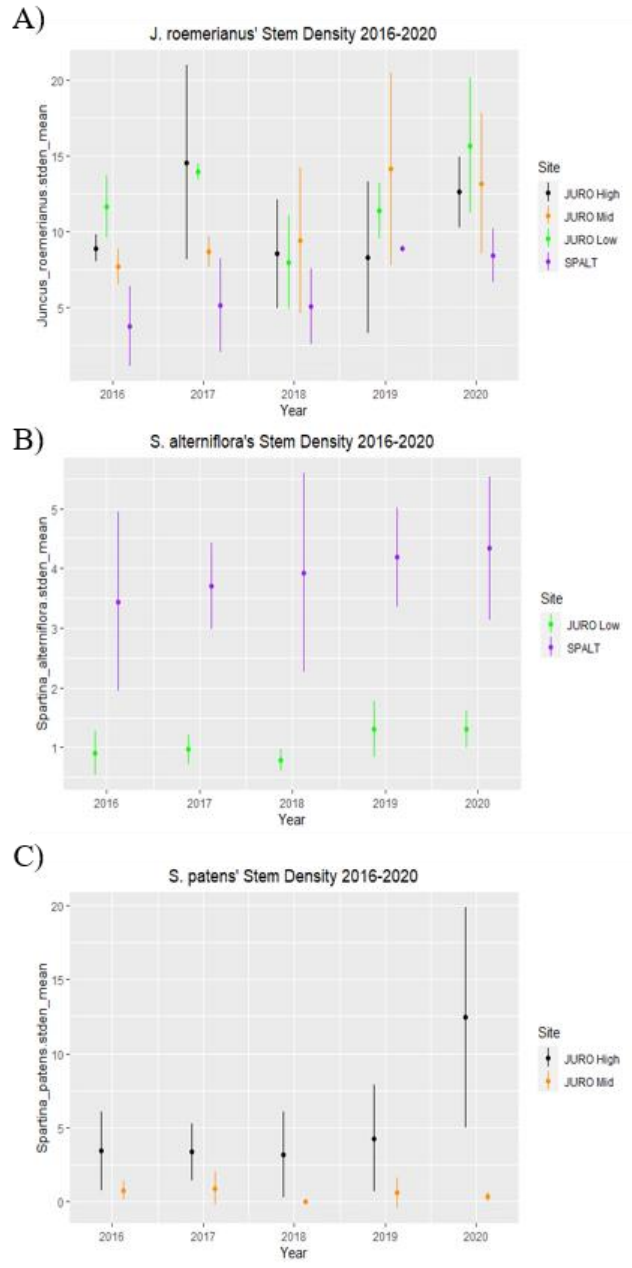


Figure 1.6 Stem density changes for *J. roemerianus*, *S. alterniflora*, and *S. patens* from 2016 to 2020 at each study site (panels A, B, C, respectively)

Each point represents the mean of the three transects and the error bars represent mean \pm standard deviation. Each site is color coded whereby JURO High is black, JURO Mid is dark orange, JURO Low is green, and SPALT is purple.

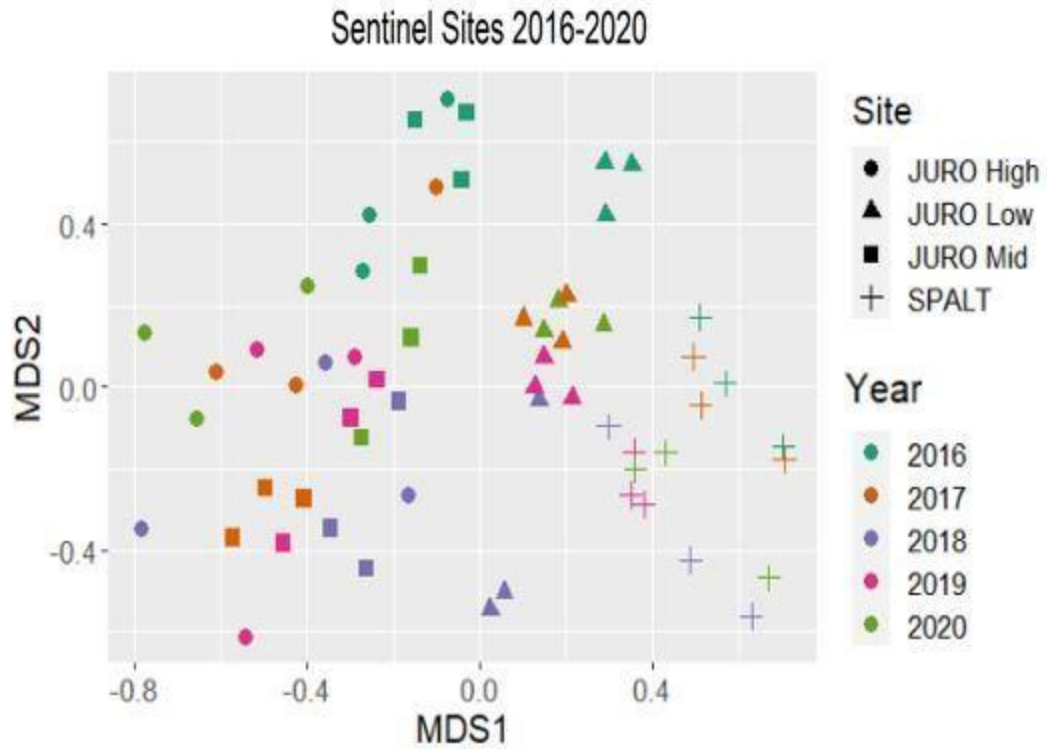


Figure 1.7 Non-metric multidimensional scaling (nMDS) of all study sites from 2016 to 2020

Each year is denoted by color and sites are abbreviated and denoted by symbology: JURO High (circle), JURO Low (triangle), JURO Mid (square), and SPALT (plus).

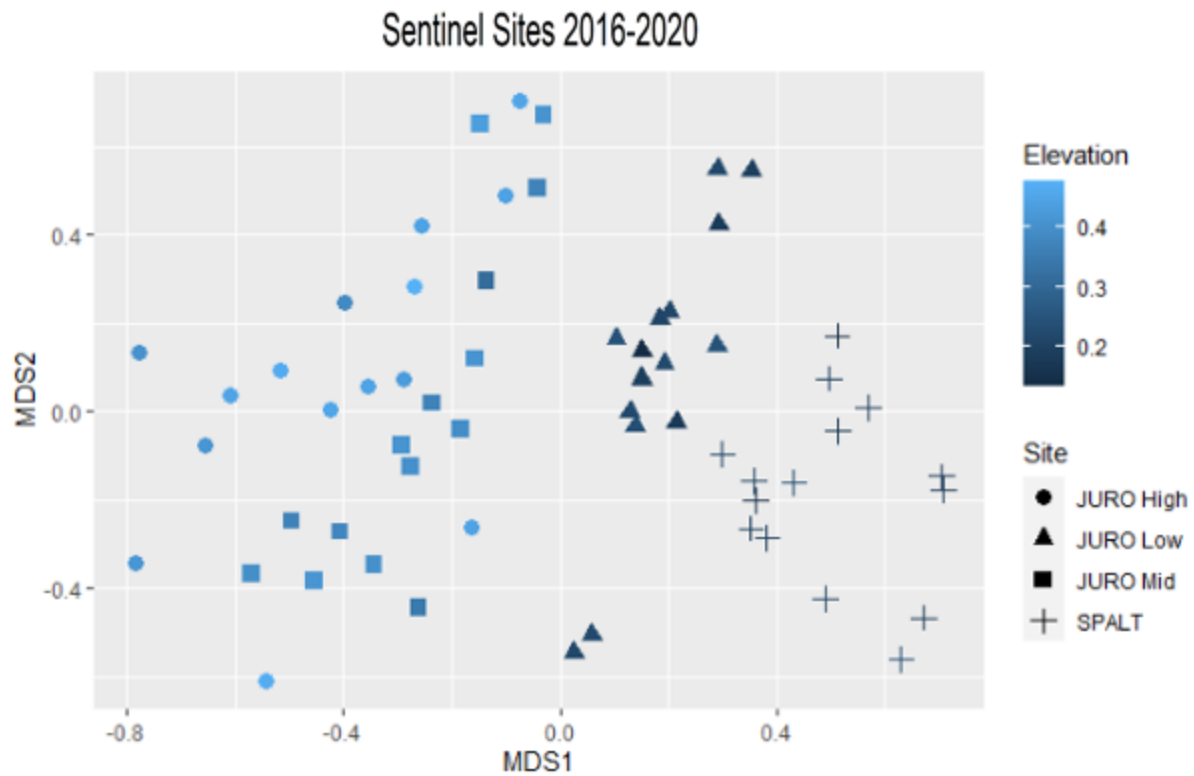


Figure 1.8 Non-metric multidimensional scaling (nMDS) of all study sites and their elevation ranges

Sites are abbreviated and denoted by symbology: JURO High (circle), JURO Low (triangle), JURO Mid (square), and SPALT (plus). Each of the points are also colored by their ranges in elevation: lower elevations are darker colors while higher elevations are lighter colors.

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