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Decomposition Rates of Spartina alterniflora (Smooth Cordgrass) in Natural and Created Salt Marshes in Coastal Louisiana

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Decomposition Rates of *Spartina alterniflora* (Smooth Cordgrass) in Natural and Created Salt Marshes in Coastal Louisiana

by

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This thesis project is dedicated to the memory of Katelyn June Lamb, an incredible mentor, researcher, student, and friend.

Abstract

The combined effects of sea level rise, subsidence, and the leveeing of the Mississippi River have led to a land loss crisis in coastal Louisiana. One solution to this problem is the creation of man-made marshes using dredged sediments in places where natural marshes once thrived. However, created and natural marshes can differ in their physical and hydrological characteristics, leading to differences in ecological functions. The aim of this study is to compare organic matter decomposition rates, one important ecological function in wetlands, among natural and created brackish marshes using a common litter bag technique. The project focuses on four sites near Port Sulphur, Louisiana, with two sites being recently created marshes. Replicate litter bags containing a standard amount of the common marsh grass *Spartina alterniflora* leaf litter were placed at plots along 100-meter transects at each site for two-month periods during the summers of 2018 and 2019. Leaf litter was dried, weighed, and compared to initial masses to estimate litter loss during this time. Litter loss and decomposition rate coefficients were compared between sites and years, with and without edge plot data, and relative to site and plot level flooding and elevation. It was determined that flooding and elevation have the greatest impact on decomposition rates. The site with the greatest elevation consequently experienced the least amount of flooding, leading to lower litter loss and decomposition rates. This information can be used when designing created marshes in the future, as we can control the elevation of sites to make them more similar to natural ones or even more resilient.

Introduction

Louisiana is currently in the middle of a land loss crisis. Between 1985 and 2010 coastal Louisiana has lost about 43 square kilometers of wetlands per year on average (Couvillion et al. 2011). In total, Louisiana has lost as much as 5,000 square kilometers of land in the last century (Jankowski et al. 2017). Without action, it is estimated that Louisiana could lose over 5,000 more square kilometers of land in the next 50 years (CPRA 2017).

Land loss is caused by a combination of factors, including subsidence, leveeing of the Mississippi River, sea-level rise, major storm events, the dredging of canals (Kennish 2001, Penland and Ramsey 1990). Subsidence occurs when sediments are compacted over time, causing the land to sink. This is especially problematic in Louisiana's wetlands, as coastal land is subsiding at a rate of about 9 mm per year (Nienhuis et al. 2017). Additionally, levees built along the Mississippi River have kept sediment from flowing into the wetlands, which has exacerbated the effects of subsidence (Kennish 2001). Sea-level rise further compounds these effects, with the best-case scenario for global sea-level rise at an increase of 0.55 meters by 2100 (Hinkel et al. 2015). Finally, major hurricanes and the dredging of canals for navigation and oil and gas development has led to significant erosion of coastal wetlands in Louisiana (Kennish 2001, Palaneasu-Lovejoy et al. 2013).

Land loss is a critical issue because coastal wetlands provide both societal and ecological services. For example, the benefits of wetlands provided through commercial fisheries, trapping, recreation, and storm protection are estimated to be worth approximately $2,429 per acre (1983 prices) at their lowest value (Costanza et al. 1989). In addition, with each mile of lost coastline an additional $5.75 million (1997 prices) in storm related damage is expected each year in Louisiana (Barbier et al. 1997). From an ecological perspective, coastal wetlands in Louisiana act as an important nursery and feeding ground for aquatic organisms and serve as an important winter habitat for migratory waterfowl (CPRA 2017).

The Coastal Protection and Restoration Authority (CPRA) of Louisiana has implemented a $50 billion dollar master plan to protect and restore coastal lands. Marsh creation projects comprise over 17 billion dollars of these efforts (CPRA 2017). However, to be successful, marsh creation projects should be designed to restore and enhance the societal and ecological services provide by these habitats. Coastal wetlands filter out chemicals from water, like heavy metals or nutrients from runoff (Assessment M. E. 2005), and they also help reduce the magnitude of hurricane storm surges and flooding (Conservation et al. 1998). The decomposition of vegetation in a wetland is another important ecological process.

When organic matter decomposes in a wetland, this allows for the cycling of useable nutrients throughout the ecosystem and for the accumulation or loss of soil (Newman et al. 2001). The presence of litter at the base of a wetland also helps spur important aerobic and anaerobic decomposition processes (Marinucci 1982). Furthermore, wetlands are some of the best ecosystems for sequestration of carbon, as the slowly decomposing vegetation contains substantial amounts of carbon that are recycled into the soil (Adhikari et al. 2009).

There are many factors that affect the rate at which wetland litter decomposition occurs. Chemically, the availability of nutrients, substrate content, and salinity can contribute to how fast or slow litter decomposes (Debusk and Reddy 2005, Stagg et al. 2018). Biological factors that influence decomposition rate are differences in wetland plant communities and the amount and diversity of invertebrates in the soil (Stagg et al. 2018, Wu et al. 2009). Physical factors, like flooding and elevation, can also influence wetland litter decomposition (Janousek et al. 2017). Understanding the importance of wetland vegetation decomposition, and the factors that impact the rate at which it occurs, can be critical for coastal and wetland restoration. Litter decomposition in wetlands, and its role in creating soil organic matter, can be different in natural and restored marshes, and more research must be done to better understand what this means for coastal restoration (Ballantine and Schneider 2009).

Because the decomposition rate of vegetation is a key ecological function of marshes, this study seeks to assess the difference in decomposition rate of *Spartina alterniflora* (Smooth Cordgrass) between created and natural marshes in coastal Louisiana. To determine this, a common litter bag technique was used at two created marsh sites and two natural marsh sites. This study can allow us to understand 1) if and how litter loss and decomposition rates differ between created and natural marshes and between years, and 2) the influence of marsh flooding on litter loss and decomposition rates.

Methods

Study Area Description

This study was conducted in 2018 and 2019 at four marsh sites, two created and two natural reference marshes, in Barataria Bay, southeastern Louisiana near Lake Hermitage and the West Pointe a la Hache siphon. The Lake Hermitage region is significant because approximately 38 percent of the original marsh was lost between 1932 and 1990 (CPRA 2020). Data from the United States Geological Survey (USGS) suggests that this project area had a yearly loss rate of about -1.64 percent between 1985 and 2006 (CPRA 2020). Because of this, Louisiana's Coastal Protection and Restoration Authority (CPRA) included this area as part of its 17 billion dollars allocated to marsh creation projects in Louisiana's Coastal Master Plan (CPRA 2017).

The two created sites analyzed in this study are associated with the project known as the "Lake Hermitage Marsh Creation (BA-0042) Project," which was funded with over 34 million dollars and implemented by the Coastal Wetlands Planning, Protection and Restoration Act (CWPPRA). The first created site, known as Lake Hermitage A (LHA; 29.552052°/-89.851797°) for this study, was constructed between August 2012 and October 2013 and approximately 6 years old at the time of the study (Fig.1). The second created site, known in this study as Lake Hermitage B (LHB; 29.552842°/-89.858334°), was built between December 2013 and May 2014 and approximately 5 years old at the time of the study (Fig. 1). Both sites were created with dredged materials from the Mississippi River (CPRA 2020).

Natural reference marsh sites include Lake Hermitage C (LHC; 29.556246°/ -89.858354°), a remnant marsh located adjacent to LHA (Fig. 1). The second natural reference marsh selected, West Pointe a la Hache 2 (WPH2; 29.515464°/ -89.787971°), was adjacent to the Louisiana's Coastwide Reference Monitoring System (CRMS) CRMS3680 station (29.512346°/ -89.791725°, Fig. 1). Like the created sites, these two reference marshes are considered brackish marshes and are dominated by *Spartina alterniflora* (Smooth Cordgrass).

Litter Bag Creation, Deployment, and Recovery

Litter decomposition was quantified using the common litter bag technique (Hackney and De la Cruz 1980). Dried *Spartina* litter was acquired from the Louisiana Universities Marine Consortium (LUMCON) in Cocodrie, Louisiana and came from live plants in the nearby salt marshes. The litter material was comprised of *Spartina* leaves and stems and was separated into sets of 25 grams weighed to the nearest 0.1 mg that consisted of 5 bundles of vegetation zip tied together. The 25 grams of vegetation were then placed into the 15 by 30-centimeter mesh bags and sewn shut. Five bags were then tied together, a meter between each bag. Each set of five bags was deployed at plots along a single transect at each of the four sites. For LHA, LHB, and WPH2, plots were 1 meter, 10 meters, 25 meters, 50 meters, and 100 meters from the marsh edge. LHC plots ended at 50 meters because this site's marsh area was not large enough to include a100 meter plot.

Litter bags were left at each site for approximately two months in each year. In 2018, litter bags were deployed at LHA and LHB on May 22, at LHC on May 23, and at WPH2 on May 21. Litter bags were recovered from all sites on July 24, 2018. Therefore, in 2018 litter bags were deployed at LHA and LHB for 63 days, at LHC for 62 days, and WPH2 for 64 days, respectively. In 2019, litter bags were deployed at LHA, LHB, and LHC on May 21 and at WPH2 on May 20. Litter bags were recovered at LHA, LHB, and WPH2 on July 22, 2019 and at LHC on July 23, 2019. Therefore, in 2019 litter bags were deployed at LHA and LHB for 62 days and at LHC and WPH2 for 63 days. Recovered litter bags were frozen in a lab freezer at negative 20 degrees Celcius until they were individually processed.

Sample Processing and Decomposition Calculations

Litter bags were then thawed and the litter material cleaned in a sink over a sieve (125 microns). Any sediment or organisms collected on the litter while in the marsh were separated so that only the plant material remained. The recovered, cleaned plant material was then placed in a drying oven at 60 degrees Celsius till constant mass (approximately five days), and then weighed to the nearest 0.1 mg using an analytical balance. Following the methods of Wu et al. (2017), I calculated litter loss (g d^{-1}) during the study period as the difference in deployment and recovery litter mass relative to the deployment duration. Additionally, I determined the decomposition coefficient (k) calculated based on an exponential decay function (Janssen, 1984):

$$k = \frac{1}{\Delta t} \ln \frac{y_t}{y_t + \Delta_t}$$

where k is the decomposition coefficient or decay constant, Δ_t is the experiment time interval (day), y_t is the initial litter weight, and $y_t + \Delta_t$ is the litter weight after Δ_t incubation period. A higher k is associated with a greater decomposition or a greater loss of litter per unit time.

Elevation Data Collection

Position and elevation were measured at each plot at each site in July 2018 and 2019 during litter bag recovery. These data were measured with real-time network (RTN) surveys using a Trimble GNSS Receiver (Model R8), Data Collector/Controller (Model TSC2), and Cell Modem (Model TDL). This was done with support from the LSU Center for GeoInformatics and Louisiana Spatial Reference Center. Position and elevation data was collected in duplicate from the center of each plot and referenced to datum NAD83 (latitudinal and longitudinal position) and NAVD88 (geodetic elevation) using the geoid model GEOID2B and projection Louisiana South (a Lambert 2-parallel projection). Vertical precision based on repeated measurements of each plot was 0.0216 m.

Hydrology and Flooding Index

Throughout the duration of litter bag deployment, hourly water levels were monitored using data obtained from the Louisiana's Coastwide Reference Monitoring System (CRMS). Specifically, hourly water level data was collected from the CRMS3680 station water level recorder, which is located near the WPH2 site, and converted into centimeters (NAVD88). Using these data, I calculated a plot level index of marsh flooding. Specifically, I compared hourly water level relative to elevation at each plot. When the water level was 1 cm or more above a specific plot's elevation, then I considered it flooded for that hour. I then calculated average hours flooded per day for each plot during the period of litter bag deployment.

Data Analysis

I used separate Analysis of Variance (ANOVA) at each site to compare litter loss (g d^{-1}) and decomposition coefficients (k) between plots located 1m from the marsh edge relative to those in the marsh interior (10 to 100m from the marsh edge). I also used separate ANOVA to test for differences in litter loss (g d^{-1}), decomposition coefficients (k), marsh elevation (m), and marsh flooding (mean hr flooded d^{-1}) among sites (LHA, LHB, LHC, and WPH2) and between years (2018 and 2019). Tukey HSD post-hoc tests were used to identify significant differences among sites and years. Given the potential for decomposition dynamics to differ between the marsh edge and platform (Rippel et al. 2020), ANOVAs with litter loss and decomposition coefficient as the response variable were conducted with and without data from edge (1 meter) plots.

In addition, I further quantified the influence of marsh flooding on litter decomposition using regression analyses. I compared separate linear and logarithmic regressions with litter loss (g d^{-1}) or decomposition coefficient (k) as the response variable and marsh flooding (mean hr flooded d^{-1}) as the predictor variable. Similar to ANOVAs, regression analyses were conducted with and without data from edge (1 meter) plots. Adjusted R^2 and Akaike's Information criterion (AIC) were used to compare predictive power between linear and logarithmic models. The model with the highest Adjusted R^2 were interpreted as having higher predictive power and those with the lowest AIC values were interpreted as those with the strongest support (Burnham and Anderson 2003). All analyses were conducted in program R version 3.6.2 (RStudio Team, 2020).

Results

Decomposition

Litter loss (g d^{-1}) was higher at plots 1m from the marsh edge relative to plots in the marsh interior that were 10 to 100m from the marsh edge at LHA ($p < 0.001$, df = 4, f = 10.830; Fig. 2), LHB ($p < 0.001$, df = 4, f = 8.566; Fig. 3), LHC ($p < 0.001$, df = 3, f = 4.835; Fig. 2), but not at WPH2 ($p < 0.088$, df = 4, f = 2.196; Fig. 2). Litter loss also differed by site ($p < 0.001$, df = 3, f = 7.663) and by year (p = 0.005, df = 1, f = 8.172). Specifically, litter loss was significantly lower at LHA relative to LHB and WPH2, while LHC and LHA had similar rates of litter loss (Table 1; Fig. 3). In addition, litter loss across all sites was significantly higher in 2019 (0.2877±0.0349 g d^{-1}) relative to 2018 (0.2720±0.0441 g d^{-1}). When excluding marsh edge plots, litter loss also differed by site ($p < 0.001$, df = 3, f = 13.84) and by year ($p < 0.001$, df = 1, f = 12.98). Specifically, litter loss was lower at LHA relative to all other sites and was higher in 2019 (0.2793±0.0310 g d^{-1}) relative to 2018 (0.2608±0.0392 g d^{-1}) across all sites (Table 2; Fig. 3).

Decomposition coefficient (k) was higher at plots 1m from the marsh edge relative to plots in the marsh interior that were 10 to 100m from the marsh edge at LHA ($p < 0.001$, df = 4, f = 14.300), LHB ($p < 0.001$, df = 4, f = 11.710), LHC ($p < 0.001$, df = 3, f = 5.502), but not at WPH2 ($p < 0.085$, df = 4, f = 2.191; Fig. 2). Decomposition coefficient also differed by site (p = 0.002, df = 3, f = 5.030) and by year (p = 0.044, df = 1, f = 4.113). Decomposition coefficients were lower at LHA relative to LHB (Table 1; Fig. 3). In addition, decomposition coefficients across all sites were higher in 2019 (0.0211±0.0053 d^{-1}) relative to 2018 (0.0194±0.0062 d^{-1}). When excluding marsh edge plots, decomposition coefficients also differ by site ($p < 0.00$, df = 3, f = 11.74) and by year (p = 0.002, df = 1, f = 10.42). Decomposition coefficients were lower at

LHA relative to all other sites and was higher in 2019 (0.0197±0.0041 d^{-1}) relative to 2018 (0.0176±0.0044 d^{-1}) across all sites (Table 2; Fig. 3).

Elevation

When including edge plots, it was determined that mean elevation (meters) did differ by site ($p < 0.001$, $df = 3$, $f = 17.84$) and by year ($p = .00745$, $df = 1$, $f = 7.32$). The mean elevations of LHB, WPH2, and LHC were found to be similar, while the mean elevation of LHA was significantly different from the other sites (Table 1, Table 2). There were also differences in mean elevation between years (Table 1, Table 2). When excluding edge plots, mean elevation did differ by site ($p = < 0.001$, $df = 3$, $f = 100.12$) and by year ($p = 0.00756$, $df = 1$, $f = 7.34$). There were significant differences in mean elevation across sites, with LHB and LHC being most similar and WPH2 and LHA being significantly different from other sites. LHA had the highest elevation across both years. There were also significant differences in mean elevation between years, with sites showing higher elevation in 2019 (Fig. 4).

Hydrology and Flooding Index

Water temperature at the CRMS3680 station averaged 29.9±1.7°C in 2018 and 30.2±1.8°C during litter bag deployment in 2018 and 2019, respectively. Salinity at the CRMS3680 station ranged from 8.3 to 13.0 ppt in 2018 (average: 10.9±1.1 ppt) and between 6.5 to 10.2 ppt in 2019 (average: 8.4±0.8 ppt). Water level at the CRMS3680 station ranged from 0.23 to 2.16 ft in 2018 (average: 11.06±0.33 ft) and between 0.10 to 4.87 ft t in 2019 (average: 1.47±0.53 ft).

When comparing flooding indices including edge plots, it was determined that the average daily hours flooded differed by site ($p < 0.001$, $df = 3$, $f = 21.52$) and by year ($p < 0.001$, df = 1, f = 32.95; Table 1). Similarities were found in average daily hours flooded between LHC, LHB, and WPH2, and that the flooding index for LHA was significantly different from the other sites (Fig. 4). LHA saw the lowest amount of flooding, while LHC saw the highest amount of flooding. There were also significant differences in flooding indices between years (Figure 4). Flooding was higher at all sites in 2019. When excluding marsh edge plots, it was determined that the average daily hours flooded did differ by site ($p < 0.001$, df = 3, f = 146.7) and by year ($p < 0.001$, df = 1, f = 105.8; Table 2). There were significant differences in flooding indices across sites, with LHB and LHC being most similar and WPH2 and LHA being significantly different from other sites (Tables 1, Table 2). There were also significant differences in flooding indices between years (Fig. 4).

Effect of Flooding on Decomposition

When examining all plots, there was a significant positive linear relationship between litter loss (g d^{-1}) and marsh flooding (hours flooded d^{-1}; Table 3). In addition, the logarithmic regression between litter loss and marsh flooding was significant, with increased litter loss at higher durations of flooding (Fig. 5). Adjusted R^2 values were higher (0.112 vs. 0.075) and AIC values were lower (-698.4 vs -690.7) in logarithmic relative to the linear regressions (Table 3). When excluding marsh edge plots, both linear and logarithmic regressions between litter loss and marsh flooding were significant, with increased litter loss at higher durations of flooding (Fig. 3). Moreover, there were higher adjusted R^2 values (0.332 vs. 0.180) and lower AIC values (-632.9 vs -593.3) found in logarithmic relative to linear regressions (Table 3). In addition, regressions conducted after excluding marsh edge plots had higher adjusted R^2 values relative to regressions conducted using all plots (Table 3).

When including all plots, there was a significant positive linear relationship between decomposition coefficient (k) and marsh flooding (hours flooded d^{-1}; Table 3). In addition, the logarithmic regression between decomposition coefficient and marsh flooding was significant, with higher k values at higher durations of flooding (Fig. 6). Adjusted R^2 values were higher (0.033 vs 0.028) and AIC values were lower (-1416.516 vs -1415.713) in logarithmic relative to the linear regressions (Table 3). When excluding marsh edge plots, both linear and logarithmic regressions between decomposition coefficient and marsh flooding were significant, with higher k values at higher durations of flooding (Fig. 3). Moreover, there were higher adjusted R^2 values (0.259 vs. 0.145) and lower AIC values (-1244.208 vs. -1222.871) found in logarithmic relative to linear regressions (Table 3). In addition, regressions conducted after excluding marsh edge plots had higher adjusted R^2 values relative to regressions conducted using all plots (Table 3).

Discussion

I quantified the decomposition rate of *Spartina alterniflora* in created and natural brackish marshes in coastal Louisiana. As a key ecological function in wetlands, understanding if and how the decomposition rate of *Spartina* differs between created and natural marshes can aid in designing sustainable marsh creation projects to combat the land loss crisis in coastal Louisiana. I found that the created site LHB had similar decomposition rates to natural reference sites LHC and WPH2, and created site LHA differed from all three of these sites. This suggests that whether a march site is created or natural is not a major factor in determining decomposition rate per se. Instead, site-specific and inter-annual differences in both years decomposition rate was higher at edge plots than in interior plots.

I found that decomposition rate differed within sites and between years, likely linked to changes in the physical characteristics of the marshes. I found that elevation and hydrological conditions differed among sites. For example, the created marsh LHA had a significantly higher elevation than the other three marsh sites. In addition, there was a slight increase in three (LHA, LHB, WPH2) out of four sites between 2018 and 2019. This indicates that accretion processes like sediment deposition and organic matter accumulation likely had higher rates than subsidence, such that there was an increase in marsh elevation at these sites (Moore et al. 2021). One important event that could have influenced differences between years was the passing of Hurricane Barry over all four marsh sites in mid-July of 2019, approximately two weeks before litter bag recovery and elevation surveys. Despite their destructive power, hurricanes can also bring sediment to coastal marshes that enhances vertical accretion and thus increases marsh elevation (Cahoon et al. 1995; McKee and Cherry 2009). In addition, other storms throughout the year may have also deposited sediment and increased elevation (Reed 1989).

In addition to sediment deposition, storm events can also increase flooding of coastal marshes. Despite having higher elevations in 2019, all sites experienced a greater average of daily hours flooded. A previous study in the coastal marshes of southern New Jersey found that increased flooding is associated with increased litter moisture and higher decomposition rates in *Spartina* spp. (Halupa and Howes 1995). In addition, studies in China and Slovenia have found higher decomposition rates of *Phragmites australis* litter in more inundated environments (Zhai et al. 2021; Dolinar et al. 2016). In my study, the most highly elevated site, LHA, had the lowest amount of average daily hours flooded and the lowest decomposition rate across both years (Table 1 and 2). The other three sites (LHB. LHC, WPH2) had lower elevations, greater flooding durations, and higher decomposition rates across both years. This suggests that differences in elevation and inundation at the site level strongly influence *Spartina alterniflora* litter decomposition across created and natural brackish marshes in coastal Louisiana. In addition, the significant relationships found between litter decomposition and flooding duration at the plot level further support this conclusion (Fig. 2 and 3).

Litter decomposition also differed within sites. I found higher litter decomposition rates in edge plots (1m) relative to interior plots (10, 25, 50, 100m plots) at all sites. This result is supported by past studies that also found differing decomposition dynamics between the marsh edge and interior. For example, *Spartina alterniflora* live roots and rhizomes placed into buried litter bags at sites in Rhode Island decomposed more rapidly on the marsh edge in comparison to the marsh interior (Ellison, Bertness, and Miller 1986). In addition, marsh edges have been found to have less litter accumulation, lower elevations, and higher decomposition rates of *Spartina patens* (Saltmeadow cordgrass) litter relative to marsh interiors in mid-Atlantic New Jersey coastal wetlands (Rippel et al. 2020). Even so, in this study I found a stronger relationship between litter decomposition and flooding duration when excluding edge plots (Fig. 3). This suggests that elevation and inundation are not the sole controls on litter decomposition along marsh edges. Other unique characteristics of marsh edges, such as wave exposure, microclimate, vegetation communities, and soil characteristics likely interact with elevation and inundation leading to additional variation in litter decomposition (Roner et al. 2016, Rippel et al. 2020).

Variation in the decomposition of *Spartina alterniflor*a litter has the potential to affect available resource availability for marsh consumers and sediment carbon sequestration (Adhikari et al. 2009, Roner et al. 2016). For example, decomposition rate and aboveground plant and litter biomass were measured as key contributors to variation in soil development across 30 restored palustrine depressional wetlands located in central New York (Ballantine and Schneider 2009). My study found that *Spartina alterniflor*a litter decomposition differed among and within four created and natural reference marshes, with differences in elevation and inundation acting as a driving factor. These sites and other marsh creation projects can play a significant role in combating land loss in Louisiana. Thousands of acres of coastal wetlands have already been restored in Louisiana since the conception of the first Coastal Master Plan in 2007 (Restore the Mississippi River Delta 2017). Therefore, when seeking to design created marshes that are functionally equivalent to natural reference marshes, especially when considering vegetation decomposition rates, elevation and inundation and two key factors that need to be taken into account.

For example, designing marsh creation project, there are factors that can and cannot be controlled that have major impacts on decomposition rates. Inundation is a factor that we cannot control per se, as it can vary from site to site and from year to year. This is especially relevant as climate change driven sea level rise and increases in storm frequency and intensity become more prominent. Marshes are at the mercy of the weather and tides when it comes to inundation, but to combat this, it is possible to control the elevation of a created marsh project. By intentionally accounting for inundation and elevation when designing a marsh project, it is possible to create marshes that are functionally equivalent to natural ones, with respect to decomposition rate of *Spartina alterniflora*. With future, long-term research about the factors that affect wetland plant decomposition, it is possible to work to create sites that are even more resilient, successful marshes that can handle changes in flooding. This will allow Louisiana’s coast to better withstand growing coastal problems for decades to come. And this can help us make a more significant impact at protecting and restoring our coast for those who depend on it.

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Tables

Table 1: Average values of litter loss, decomposition coefficient, elevation, and flooding for each site during 2018, 2019, and in total when including edge plots.

Year	Metric	Created marshes		Reference marshes	
		LHA	LHB	LHC	WPH2
2018	Litter loss (g d-1)	0.2346±0.0531	0.2888±0.0355	0.2788±0.0284	0.2873±0.0295
	Decomposition coefficient (k)	0.0152±0.0059	0.0219±0.0072	0.0195±0.0044	0.0214±0.0045
	Elevation (m)	0.1721±0.1360	0.0915±0.1213	0.0745±0.0283	0.0921±0.0232
	Flooding (hr flooded d-1)	6.91±8.57	12.12±7.27	14.08±2.37	12.67±2.12
2019	Litter loss (g d-1)	0.2862±0.0470	0.3031±0.0307	0.2787±0.0233	0.2811±0.0287
	Decomposition coefficient (k)	0.0215±0.0077	0.0232±0.0050	0.0195±0.0032	0.0200±0.0037
	Elevation (m)	0.2359±0.1042	0.1197±0.0997	0.0668±0.0782	0.1432±0.0596
	Flooding (hr flooded d-1)	9.93±6.57	17.69±6.12	20.27±2.91	16.73±3.55
All	Litter loss (g d-1)	0.2604±0.0561[a]	0.2960±0.0336[b]	0.2987±0.0256[ab]	0.2842±0.0290[b]
	Decomposition coefficient (k)	0.0183±0.0075[a]	0.0225±0.0061[b]	0.0195±0.0038[ab]	0.0207±0.0041[ab]
	Elevation (m)	0.2040±0.1242[a]	0.1056±0.1108[b]	0.0706±0.0582[b]	0.1177±0.0517[b]
	Flooding (hr flooded d-1)	8.42±7.7119[a]	14.91±7.2219[b]	17.17±4.0845[b]	14.70±3.5475[b]

Table 2: Average values of litter loss, decomposition coefficient, elevation, and flooding for each site during 2018, 2019, and in total when excluding edge plots.

Year	Metric	Created marshes		Reference marshes	
		LHA	LHB	LHC	WPH2
2018	Litter loss (g d-1)	0.2179±0.0446	0.2746±0.0200	0.2715±0.0134	0.2820±0.0280
	Decomposition coefficient (k)	0.0132±0.0045	0.0189±0.0027	0.0181±0.0017	0.0205±0.0039
	Elevation (m)	0.2348±0.0519	0.0481±0.0933	0.0715±0.0324	0.0955±0.0249
	Flooding (hr flooded d^{-1})	2.88±2.66	14.85±5.25	14.28±2.73	12.40±2.30
2019	Litter loss (g d-1)	0.2694±0.0357	0.2960±0.0287	0.2722±0.0215	0.2781±0.0297
	Decomposition coefficient (k)	0.0184±0.0045	0.0219±0.0043	0.0186±0.0028	0.0196±0.0037
	Elevation (m)	0.2859±0.0240	0.0763±0.0511	0.0485±0.0829	0.1303±0.0600
	Flooding (hr flooded d^{-1})	6.80±1.66	20.54±2.14	20.82±3.19	17.55±3.53
All	Litter loss (g d-1)	0.2436±0.0476[a]	0.2853±0.0267[b]	0.2719±0.0176[b]	0.2801±0.0286[b]
	Decomposition coefficient (k)	0.0158±0.0052[a]	0.0204±0.0038[b]	0.0184 0.0023[b]	0.0200±0.0038[b]
	Elevation (m)	0.2603±0.0476[a]	0.0622±0.0756[b]	0.0600±0.0629[b]	0.1129±0.0487[c]
	Flooding (hr flooded d^{-1})	4.84±2.9552[a]	17.70±4.8918[b]	17.55±4.4224[b]	14.97±3.9281[c]

Table 3: Logarithmic and linear regression analysis values for litter loss and decomposition coefficient for all sites including and excluding edge plots.

Response variable (y)	Data source	Regression type	Equation	Adjusted R^2	p-value	AIC
Litter loss (g d-1)	All plots	Linear	y = 0.25701 + 0.00168 x	0.075	<0.001	-690.7
		Logarithmic	y = 0.24083 + 0.01641 ln(x)	0.112	<0.001	-698.4
	Excluding 1m plots	Linear	y = 0.23846 + 0.00234 x	0.180	<0.001	-593.3
		Logarithmic	y = 0.20868 + 0.02581 ln(x)	0.332	<0.001	-623.9
Decomposition coefficient (k)	All plots	Linear	y = 0.01816 + 0.00016 x	0.028	0.011	-1415.71
		Logarithmic	y = 0.01709 + 0.00135 ln(x)	0.033	0.007	-1416.52
	Excluding 1m plots	Linear	y = 0.01523 + 0.00025 x	0.145	<0.001	-1222.87
		Logarithmic	y = 0.01213 + 0.00274 ln(x)	0.259	<0.001	-1244.21

Figures

Figure 1. Map of study area showing both natural sites (green), both created sites (yellow), and the CRMS water station (orange) and their position in coastal Louisiana.

Figure 2. Average litter loss at each plot for each site across both years

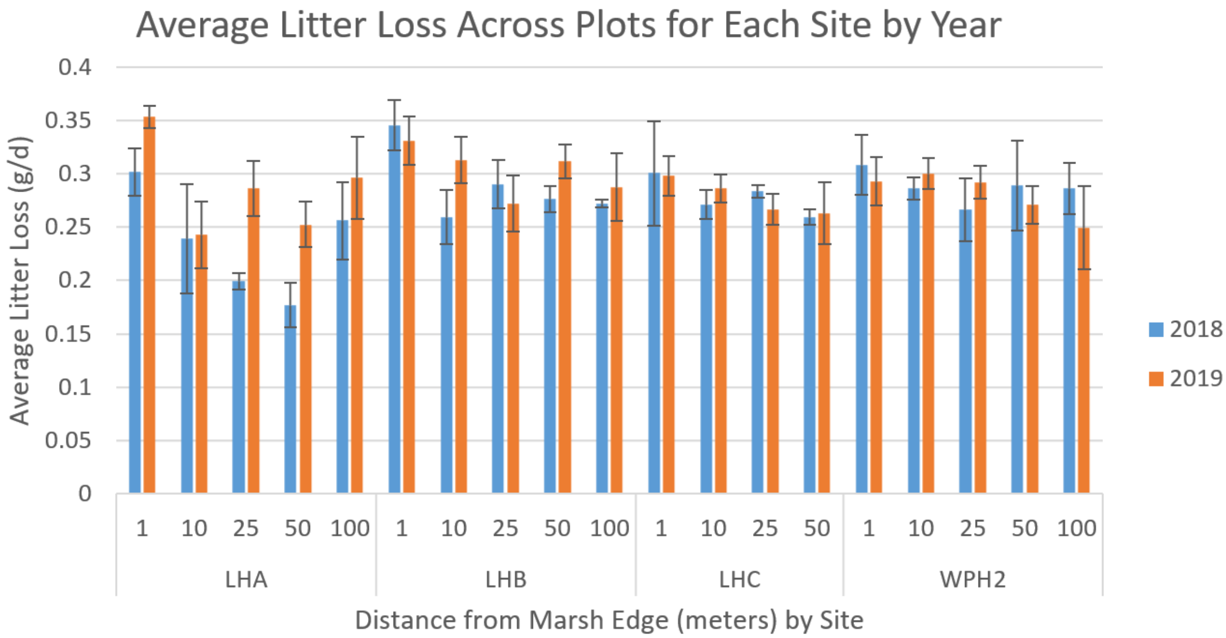
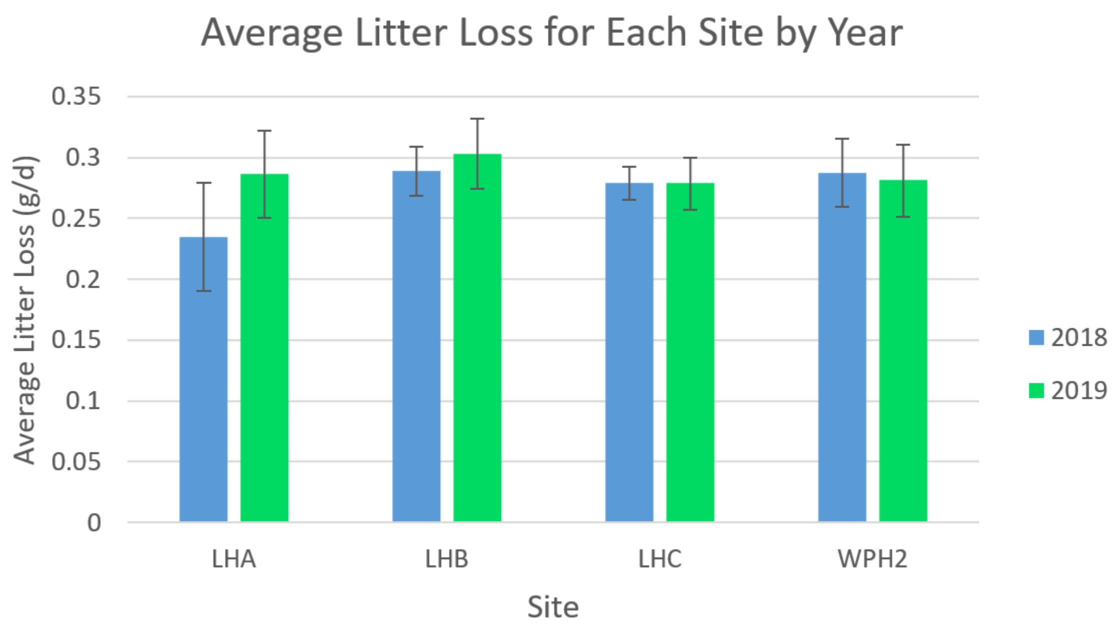


Figure 3. Average litter loss for each site including all plots across both years (top) and average decomposition coefficient for each site including all plots across both years (bottom).



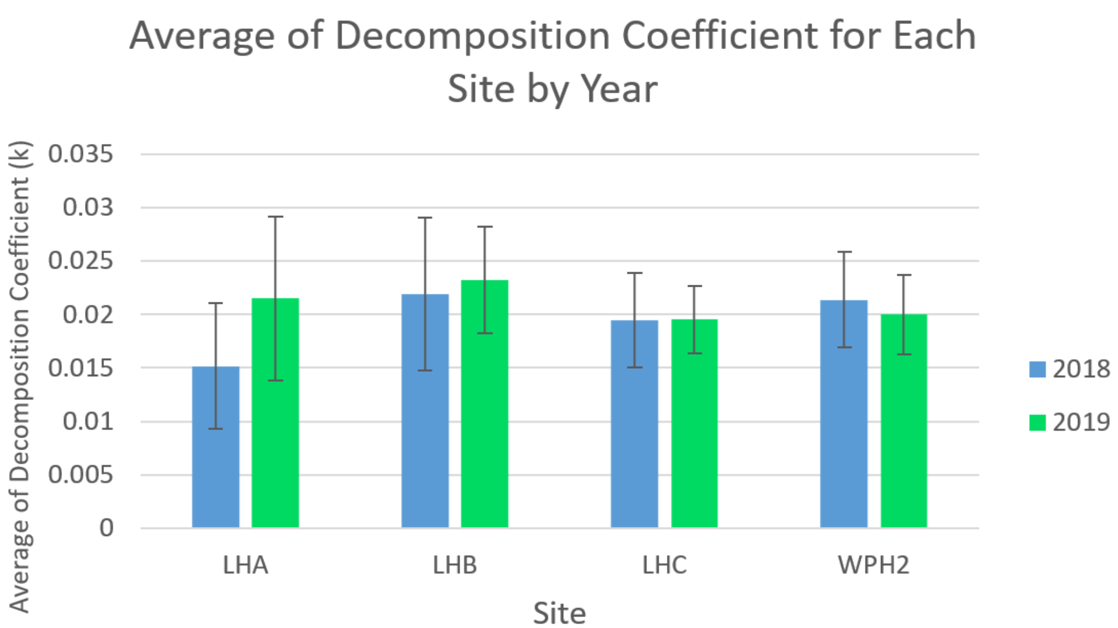
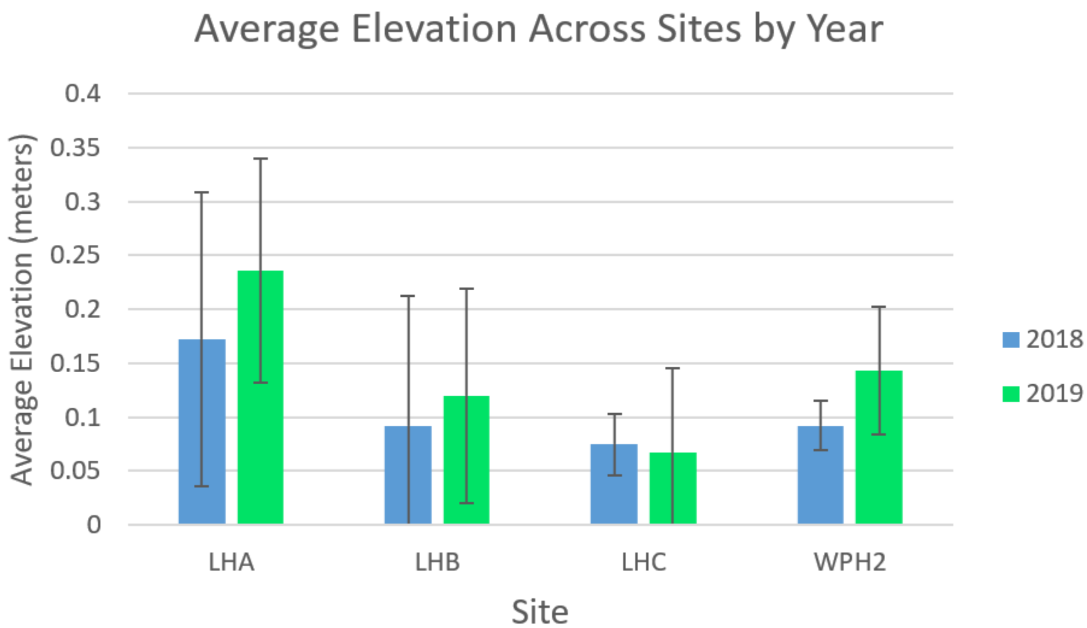


Figure 4. Average elevation across all four sites, including edge plots, by year (top) and average daily hours flooded across all four sites, including edge plots, by year (bottom).



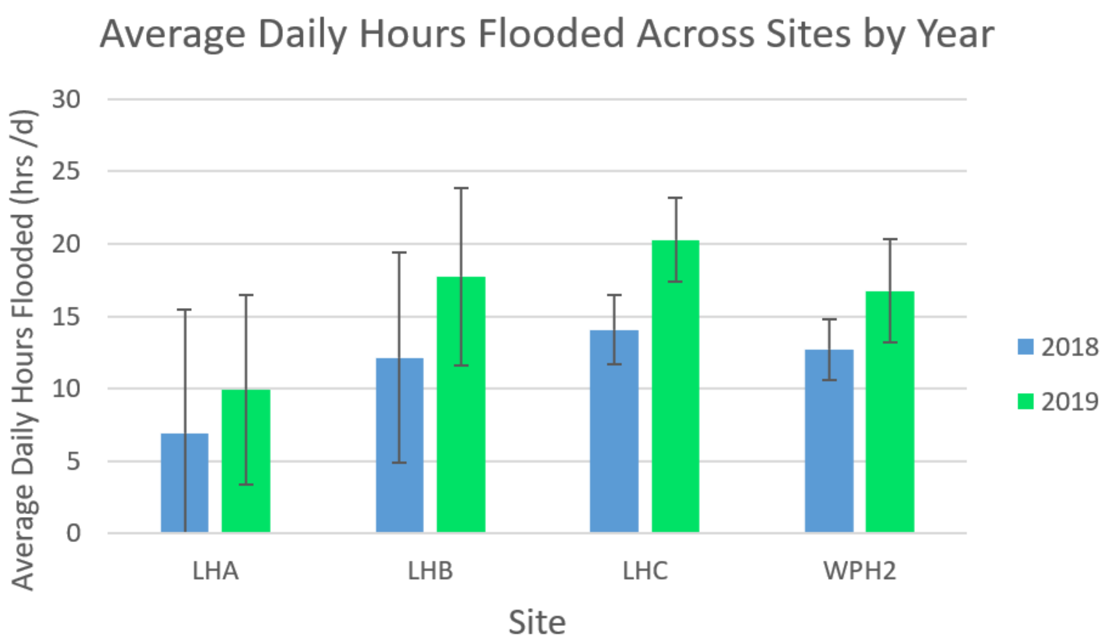


Figure 5, Logarithmic regression of litter loss for each site for all plots when compared with hours flooded per day (top) and decomposition coefficient for each site for all plots when compared with hours flooded per day (bottom).

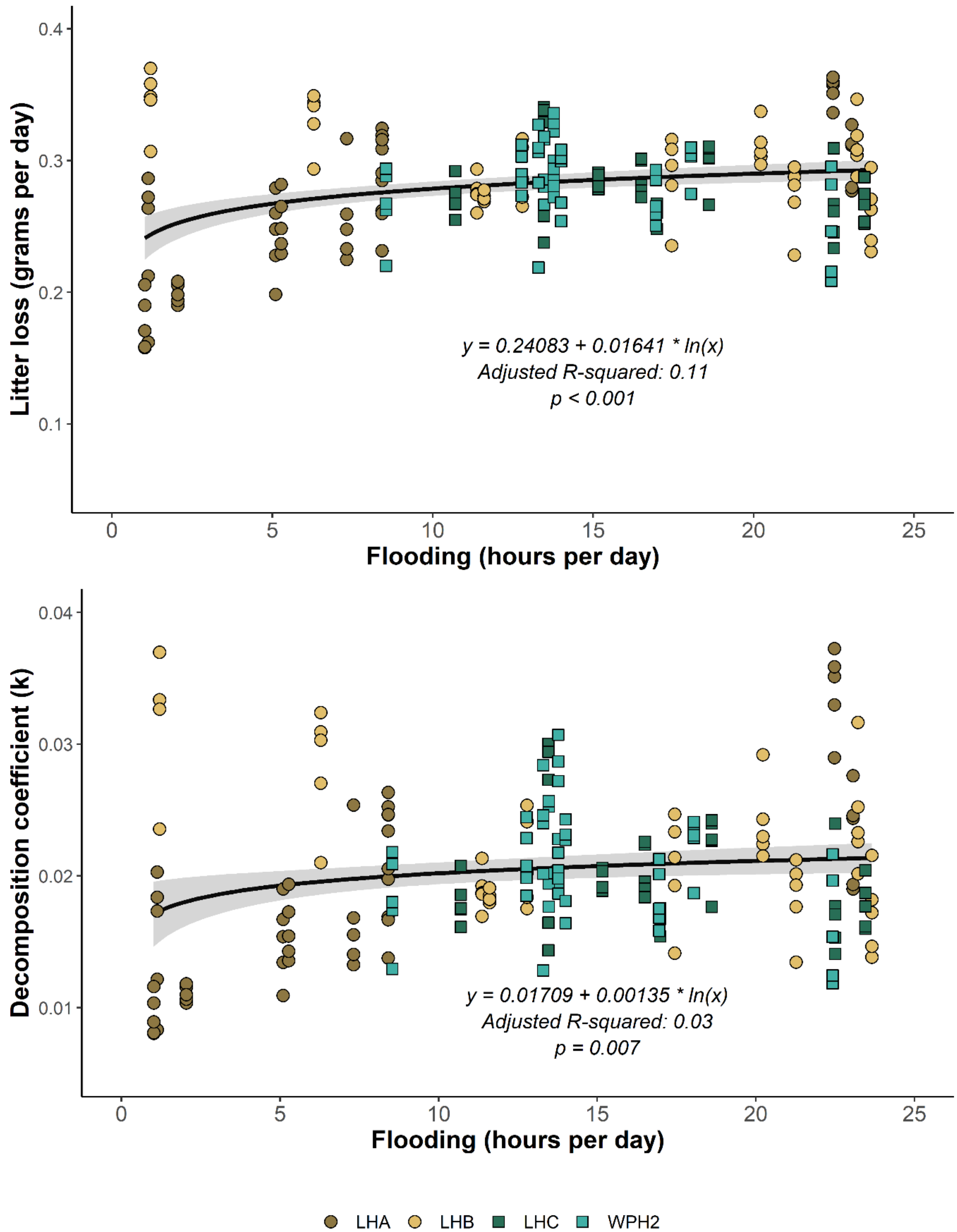


Figure 6. Logarithmic regression of litter loss for each site excluding edge plots when compared with hours flooded per day (top) and decomposition coefficient for each site excluding edge plots when compared with hours flooded per day (bottom).