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Spatial and Temporal Variability in Carbon and Nutrient Concentrations Within the North Inlet Estuary,SC

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SPATIAL AND TEMPORAL VARIABILITY IN CARBON AND NUTRIENT CONCENTRATIONS WITHIN THE NORTH INLET ESTUARY, SC

by

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ABSTRACT

Coastal wetlands provide numerous ecosystem services, including the ability to sequester and store significant amounts of organic carbon (so-called "blue carbon"). Variability in carbon storage within marshes represents a major information gap in understanding and quantifying the role saltmarshes play in the global carbon cycle. This study quantified decadal and small-scale variability in carbon and nutrient concentrations across tidal inundation gradients in the relatively pristine North Inlet Estuary, South Carolina. Sampling took place within two segments of the marsh platform of Crab Haul Creek, the landward-most basin within North Inlet. In the summer of 2020, a total of 200 sediment cores were collected to a depth of 30 cm (root zone) that spanned the elevation and vegetation community gradient from the creek bank to the upland forest edge. Cores were sectioned in 10 cm intervals and analyzed for bulk density, grain size, percent organic matter, total carbon, total nitrogen, and total phosphorus concentrations. Results from 2020 were compared to similar data collected a decade earlier at the same locations to assess the potential for temporal changes associated with increased sea level.

Despite substantial increased tidal inundation over the decade, sediment carbon and nutrient densities were not significant. In contrast, measures of sediment bulk density and carbon and nutrient concentrations had considerable small-scale variability, both between segments and across marsh elevation gradients and vegetation communities. Sediment

carbon concentrations were highly variable (0.2 to 17.1%), with younger marsh, upstream marshes characterized by ~ 2 times more carbon relative to the downstream, older marsh region. Similar to previous studies, there was a significant and strong inverse relationship (R^2 = 0.86) between sediment dry bulk density and organic carbon concentration (-0.48ln(OM) + 1.9577), which reduced spatial variability in sediment carbon densities (mean = 0.034± 0.006 g cm^{-3}). Carbon to nitrogen density ratios were relatively constant across vegetation type, marsh region, and with increasing depth, and averaged 25.52 ± 2.62.

Combined these results indicate that small-scale spatial variability within a single marsh ecosystem may be as great or greater than across marsh ecosystems. Thus, regional carbon storage requires more detailed measurements in order to scale to global estimates of the role of salt marshes in carbon storage. At the same time, strong relationships between physical parameters and the relatively low range in carbon densities, provides a mechanism for reducing the types of analyses and constraining regional estimates. While spatial variability in carbon storage is relatively stable over the roughly 150 mm of sea level rise the marsh has experienced over the past decade, it is unclear if this stability will remain under current or accelerated rates of sea level rise predicted under most recent climate change scenarios.

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CHAPTER 1: INTRODUCTION

Coastal ecosystems such as salt marshes, mangroves and seagrasses are among the most biologically productive ecosystems in the world (Howard et al., 2014). They occur globally along coastlines where saline and fresh waters mix to create ecological hotspots of diversity (Alongi, 2020). Their high rates of productivity play an important role in carbon and nutrient biogeochemistry particularly with regards to carbon sequestration and nutrient exchange with coastal waters (Mcowen et al., 2017). They also provide numerous ecosystem services including protection from storm surges and erosion by buffering wave action and trapping sediment, improve water quality by filtering excess nutrients, provide habitats for commercially important marine species, and support food security for many coastal communities around the world (Roberston and Alongi, 1992; Barbier et al., 2011; Sousa et al., 2012). Additionally, recent work has recognized that estuarine ecosystems pay a critical role in the global carbon cycle, due to their ability to sequester and store vast amounts of carbon, a reservoir that is commonly referred to as "blue carbon" (Arriloa and Cable, 2017; Spivak et al., 2019).

Blue carbon has become an increasingly important scientific and political topic that requires a better understanding of the dynamics of these natural carbon sink systems (Howard et al. 2017). Coastal ecosystems show high potential for climate mitigation, due to the large amounts of carbon that is stored in their biomass and in their soil (Hiraishi et al., 2014). For example, a salt marsh's primary production rates range from 2.96 to 6.7

kg m^{-2} y^{-1} when compared to terrestrial ecosystems, that have considerably lower production rates that range from 1.12 to 2.55 kg m^{-2} y^{-1} (Madrid et al., 2012; Duarte et al., 2013). Blue carbon ecosystems are remarkably efficient at burying organic matter with their carbon accumulation rate per unit area, 30-50 times higher than forest ecosystems. (Bridgham et al., 2006; Arriloa and Cable,

2017; Spivak et al., 2019). As such, while salt marshes may only account for 2.5% of the total global land surface area, they have a net global carbon storage of 1,400 to 1,600 Pg C which is equivalent to 20-25% of the worlds soil organic carbon (Baustian, 2017).

Understanding the magnitude of carbon storage is complicated by the large range in carbon burial potential (18-1713 g C m^{-2}) that may be a result from small-scale variability between marsh vegetation, geomorphology, and biogeochemical interactions (Arriloa and Cable, 2017). The accumulation and storage potential of organic matter may therefore be dependent on changes in environmental conditions such as salinity, temperature, nutrient input, and inundation frequency, all of which can vary across intertidal elevation gradients (Marín-Spiotta et al., 2014; Alongi, 2020). Additionally, sea level rise, subsidence, and storm surges greatly influence marsh productivity, distribution, and vertical accretion rates leaving carbon storage vulnerable to predicted changes in climate (Macreadie et al., 2019; Craft, 2012).

Sea level rise is an important driver that influences blue carbon storage and may cause salt marsh ecosystems to transgress inland, leading to a conversion of terrestrial forests into a tidal wetland environment (Craft, 2012). As sea level rises, there is increasing inundation of salt water into terrestrial environments, changing the salinity porewater concentrations and shifting microbial communities and thus the magnitude and

composition of buried organic carbon (Chambers et al., 2013). This scenario is further exacerbated by land use changes and development, referred to as the “coastal squeeze effect”, which varies with the space available for landward migration (Pontee, 2013). The various processes detailed above have made predicting changes in soil organic carbon (SOC) difficult (Spivak, 2019). Indeed, there is rising concern that anthropogenic disturbances of coastal wetlands, such as conversion and degradation, will result in a shift of these ecosystems from an atmospheric carbon sink to a carbon source (Pendleton et al., 2012). This shift would have a critical impact on the creation of climate mitigation policies and national greenhouse-gas inventories, which include CO_2 and methane (Howard et al., 2017).

The physical stability of a marsh depends on the relative elevation of sediments within the intertidal zone, as elevation influences the frequency and duration of tidal inputs and therefore vegetation productivity (Morris et al., 2002). Conditions of low to moderate rates of sea level rise can promote marsh productivity and biomass density that in turn enhances sedimentation and stability and increases carbon storage potential (Morris et el., 2016; Spivak, 2019). In contrast, rapid sea level rise can destabilize soils and expose buried organic matter that alter carbon and nutrient reservoirs (Craft, 2012; Spivak, 2019).

Currently, global sea level is expected to rise by 18 – 23 cm by the year 2050 (IPCC, 2021). However, relative rates of sea level can vary geographically and are influenced by regional factors, including subsidence, sediment supply and coastal development. These rapid rates of sea level rise may eventually lead to a reduction of suspended sediment needed for accretion and a decrease in plant productivity from

increased inundation (Schile et al., 2014). Therefore, in the absence of upslope migration (Schuerch et al., 2019), it remains uncertain whether salt marsh habitats can maintain their vegetated elevations under such rapid sea level rise and thus maintain their small-scale carbon storage capacity (Schile et al., 2014; St. Laurent et al., 2020).

The goal of this project is to understand the processes that influence sediment carbon storage by examining temporal and spatial changes within a single marsh basin. Here we: 1) assess the small-scale variability of soil carbon and nutrient concentrations 2) examine the relationship between organic matter, carbon, nitrogen and phosphorus concentrations with depth, position in the tidal frame and across vegetation communities, and 3) evaluate if soil carbon densities have changed over the past decade as a function of increased tidal inundation from rising sea levels. Results will help to determine if fine-scale variability is an important factor to consider in measuring marsh organic carbon storage, evaluate if decadal increases in inundation result in differences in sediment carbon storage capacity and allow for improved estimates of carbon storage dynamics in these critical environments.

CHAPTER 2: MATERIALS AND METHODS

2.1. STUDY AREA

The North Inlet Estuary is located along the southeastern coast of the United States and is a part of the North Inlet-Winyah Bay National Estuarine Research Reserve (NI-WB NERR), one of 30 NERR throughout the United States. North Inlet is a relatively undisturbed (by local anthropogenic factors) marsh estuarine ecosystem located near Georgetown, SC (33°20′N, 79°10′W) (Buzzelli et al., 2004; Allen et al., 2014). The region has a temperate to sub-tropical climate averaging 18°C annually with an average rainfall of 130 cm y^{-1} (Allen et al., 2014). The marsh basin spans ~ 32 km^2 and consists of 82% intertidal marsh and mudflat and 18% open water (Morris et al., 2002). The average salinity values are near oceanic at 34.6 with minimal input of freshwater from rainfall, groundwater, and surface runoff (Dame and Kenny, 1986). North Inlet is dominated by semidiurnal tides with a mean range of 1.4 m and maximum range of 2.4 m (Dame and Kenny, 1986). The marsh has evolved from a forested beach ridge terrain under slow rates of rise in sea level rise of 0.3 cm y^{-1}, whereby the invasion of salt water into low-lying swales formed new marsh basins between the forest ridges (Gardner and Porter, 2001). The landward margin of the marsh basin exhibits distinct vegetation communities that are determined by tidal inundation, groundwater salinity and plant salt tolerance and competition (Pennings et al., 2004; White and Madsen., 2016). Under gradual sea level rise conditions, the vegetation boundaries may recede inland where the

forest transforms into marsh as a result of changes in salinization and marine sediment deposition (Arriola and Cable, 2017).

The vegetation in the upland forest consists of oak and pine trees within sandy Spodosol soils (Goñi, 2000). The organic matter input within the forest sites is mainly derived from litterfall, with minimal mineral deposition (Gardner et al., 1992). Along the mean higher high water (MHHW) boundary in the high marsh, the species *J. roemerianus* and *S. europaea* (low salinity tolerance) are found. This area is inundated during spring tides which alters redox conditions and slightly increases the clay content and salinity (Gardner et al., 1992; Goñi, 2000). The mixed meadow region lies in the mid marsh zone along the MHHW mark and is also inundated during spring tides. For this study, we define the mixed meadow region to include a combination of the species *S. europaea* with vast areas of the salt panne. In North Inlet, the salt panne here is composed mainly of sand sized sediments, little to no vegetation, high salt content and poor drainage of waterlogged soils (Ewanchuk and Bertness, 2004). The low marsh consists primarily of the smooth cordgrass species *S. alterniflora* (high salinity tolerance) that varies in height from a short 0.5 m form, found within the mid to low marsh region, to a taller 1.5 m form near the creekbanks (Dame and Kenny, 1986). The *S. alterniflora* regions are inundated by spring and neap tides that supply large allochthonous inputs and silt and clay sized sediments (Gardner et al., 1992; Goñi, 2000).

Sediment cores were collected within two long-term monitoring segments (A and B), in the North Inlet Estuary, which are ~ 0.8 km apart and located along the western-most, landward tidal creek, Crab Haul Creek (Fig. 2.1). Seg. A is in a geologically older and more successionally mature portion of the marsh platform, while Seg. B, near the

head of the creek, is a relatively younger marsh (Gardner and Porter, 2001). As a result, Seg. A contains a series of vegetation transitional zones (high marsh, mid-marsh with salt panne, low marsh, tidal flat) while Seg. B has a narrow fringe of high marsh that abruptly transitions to low marsh. These two segments continue to transgress inland with gradual sea level rise (Morris et al., 2002).

Within each segment, three transects consist of 7 to 9 permanent plots that extend across the marsh platform, from upland forest edge to creekbank (Fig. 2.1). The permanent plots are 1 m^2 quadrants selected in a stratified-random manner to capture the distinct vegetation communities along the platform elevation gradient. Although the transects of both segments span roughly the same elevation gradient, a decline of approximately 1 m from the forest to creek edge, they do so over dramatically different horizontal distances (Fig. 2.2). As is typical of Southeastern salt marshes of the USA, the transition to monocultures of *S. alterniflora* in the low marsh occur approximately at the MHHW elevation. Sampling plots along the Seg. B transects are largely dominated by *S. alterniflora* communities, with the higher elevation plots occupied by a monoculture of *J. roemerianus*. In contrast, transects in Seg. A also include mid-marsh communities of mixed vegetation and salt panne (Fig. 2.1).

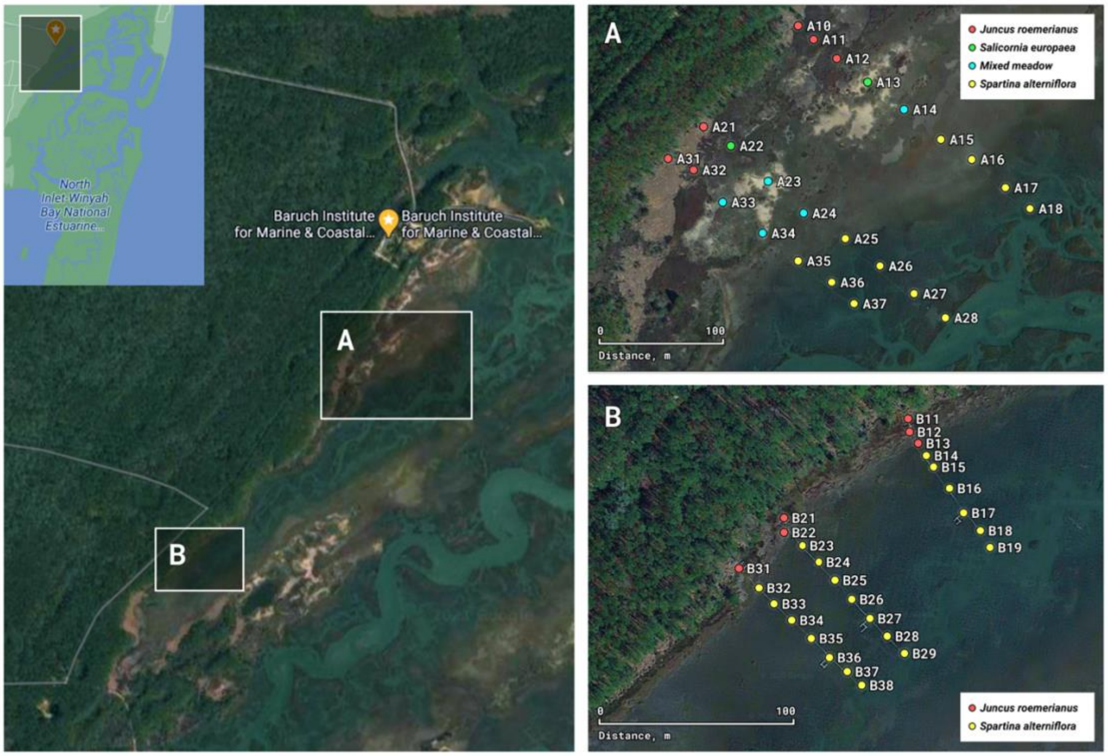

Figure 2.1 Study Area Map. Aerial map of Segment A and Segment B within the Crab Haul Creek basin of North Inlet Estuary. Each segment contains three transects and the plots are labeled and colored by vegetation communities: *Juncus roemerianus* (red circles), *Salicornia europaea* (green circles), Mixed meadow (blue circles), which is comprised of *Salicornia europaea* and the salt panne, and *Spartina alterniflora* (yellow circles).

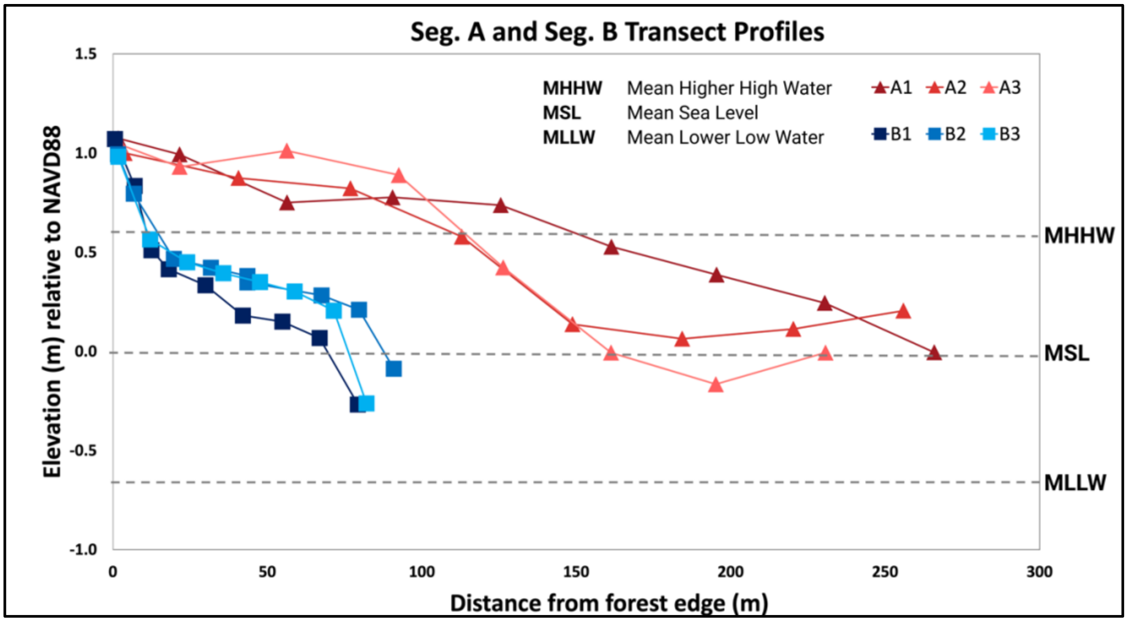

Figure 2.2 Elevation Plot of Transects. Elevations of Seg. A and Seg. B transects with distance from the forest edge. Elevation was determined by the North American Vertical Datum of 1988 (NAVD88). The mean higher high water (MHHW) mark denotes the boundary between the high and low marsh. Mean sea level (MSL) and mean lower low water (MLLW) are also indicated.

2.2. FIELD SAMPLING

Field sampling occurred in the summer of 2020 within the Crab Haul Creek basin and repeats identical sampling that occurred in 2010 by the North Inlet - Winyah Bay Nation Estuarine Research Reserve (NIWB NERR) that was never published. Dominant vegetation communities in each plot were identified and there was no change in these communities over the decade. Four sediment cores were collected at each plot using either a metal (2 cm inner diameter, 31.42 cm^3 volume) or polyvinyl chloride (PVC) corer (4.3 cm inner diameter, 145.22 cm^3 volume) depending on the soil conditions and selected to prevent significant sediment compaction during coring. At each plot, a total of 4 cores were collected using the same corer. Three of the cores were used for replicate measurements of bulk density, percent organic matter, total organic carbon, total

nitrogen, and total phosphorus concentrations. The fourth core was used for grain size analysis. All cores were collected during slack low tide under dry conditions to further reduce sediment compaction. Once collected, cores were immediately sectioned into three depth horizons (0 -10 cm, 10 - 20 cm, and 20 - 30 cm), returned to the laboratory and kept cold (4°C) until processing, as described below. The top 30 cm represents the depth of the active root zone and is typical of other salt marshes (St. Laurent et al., 2020).

2.3. LABORATORY ANALYSES

GRAIN SIZE & DRY BULK DENSITY

Grain size was determined using a Beckman Coulter LS 13 320 Particle Size Analyzer and differentiated the following size classes: %sand, %silt and %clay. For dry bulk density (DBD) measurement, a sample of known volume was weighed wet, then dried at 60°C until a constant weight was achieved (4 to 7 days) (Arriola and Cable, 2017). Estimates of DBD were then calculated as the mass of the dry solid divided by the original wet volume (Dingman, 2008).

PERCENT ORGANIC MATTER

Percent organic matter (%OM) was determined by loss of ignition (LOI) method (Craft et al., 1991). Dried sediment was first homogenized using a mortar and pestle and then using a Thomas Scientific Wiley mill grinder with a No. 20 mesh size. Sub-samples of 2 - 6 g of ground sediment was placed into pre-combusted 20 ml glass vials and ignited in a Lindberg Blue M1100 muffle furnace at 500°C for 4.5 h (Arriola and Cable, 2017). After initial combustion, samples were mixed in the vials and re-combusted to ensure complete oxidation of all organic matter.

Dry weight and ash weight were calculated as the difference between the total weight of the sample and the initial crucible weight. Percent organic matter was then calculated by the following equation (Eq. 1), where Wt_{sed} denotes the dried homogenized sediment weight and Wt_{ash} is the ash weight of the sample after combustion (St. Laurent et al., 2020).

Equation (1)

$$LOI\ (\%) = \left(\frac{Wt_{sed} - Wt_{ash}}{Wt_{sed}}\right) * 100$$

NUTRIENTS: TOTAL PARTICULATE CARBON, NITROGEN AND PHOSPHORUS

Sub-samples of dried ground sediments were analyzed for total carbon (TC), total nitrogen (TN) and total phosphorus (TP). Samples for TC and TN were stored in 20 ml glass vials and kept in a freezer at -18°C until analysis. TC and TN were analyzed on a Costech ECS 4010 elemental analyzer following the methods of Hedges and Stern (1984) without acid digestion (McCabe et al., 2021). Samples were run alongside a Consensus Reference Material (Miami, FL USA), and a subset of samples were run in duplicate with an average analytical error of < 1% coefficient of variation. (McCabe et al., 2021).

Another subset of samples (n= 72) was analyzed for percent organic carbon (%OC) after acidification with 10% hydrochloric acid. No significant difference was found between percent acidified and unacidified samples due to the negligible amount of inorganic carbon (IC) found in marsh sediments. Therefore, total particulate carbon TC is considered equal to particulate organic carbon (POC) (McCabe et al., 2021). Samples for TP were stored refrigerated at 4°C until processing for analysis. TP was analyzed according to the methods described by Aspila et al. (1976) as modified by Benitez-

Nelson et al. (2007). Samples were measured using a Beckman Coulter DU 640 Spectrophotometer at a wavelength of 880 nm.

2.4. STATISTICAL ANALYSES

Statistical analyses were conducted in R version 4.0.3. To examine temporal changes in grain size, DBD, %OM, and %OC, a Student's paired t -test was used and the statistical significance determined when the p value < 0.05. Spatial variability across segments, vegetation communities, and depth intervals for 2020 was examined. A one-way ANOVA, Mann-Whitney U test was used to determine the significant differences between Seg. A and Seg. B. A two-way ANOVA, Kruskal-Wallis post hoc Dunn test was then used to assess for spatial variability across the dominant vegetation communities and depth intervals within both segments. The R package (multcompView) was used to compare significance (p values) across the multiple groups. All tests were nonparametric due to unequal sample sizes and nonnormal distributions between groups as confirmed by a Shapiro-Wilk test.

Linear regression analyses were used to evaluate relationships between %OM and %OC as well as between TN and OC concentrations (Craft et al., 1991). Nonlinear regression analyses included natural log and second order quadratic models, to relate DBD and percent OM to percent OC and percent OM respectively. Best model fits were determined by using Pearson's product-moment correlation across all regression analyses (St. Laurent et al., 2020).

CHAPTER 3: RESULTS

3.1. GRAIN SIZE

Grain sizes were defined by %sand, %silt and %clay and ranged from 24.88 to 78.20%, 9.16 to 62.94% and 2.84 to 19.13% respectively (Table 3.1). Seg. A and Seg. B, both showed a significant ($p < 0.0001$) changes across all grain sizes from 2010 to 2020. Specifically, in Seg. A, the %sand decreased (65.64% to 59.61%), while %silt (27.46% to 31.73%) and %clay (6.91% to 8.66%) increased. In Seg. B, similar decadal changes occurred: %sand decreased (63.24% to 53.24%), while %silt (30.74% to 38.72%) and %clay (6.03% to 8.04%) increased.

Significant spatial differences were found in grain sizes across the dominate vegetation communities and depth intervals (Table 3.1). In Seg. A, %sand was greatest in the mixed meadow due to the presences of a salt panne, followed by the high marsh region that contains the species *S. europaea* (Table 3.1). The low marsh, dominated by *S. alterniflora*, had the greatest %silt (Table 1). The highest %clay was found in the upper high marsh, dominated by *J. roemerianus* (Table 3.1).

Similar trends were observed in Seg B, where the upper marsh had significantly ($p < 0.0001$) greater %sand and the low marsh had significantly ($p < 0.001$) higher %silt and %clay (Table 3.1). Although the trends were similar across segments, on average, the %sand was significantly ($p = 0.009$) higher in Seg. A relative to Seg. B, while the %silt

was significantly ($p = 0.0001$) higher in Seg. B. The %clay was not significantly ($p = 0.77$) different between the segments.

Depth intervals were also compared across vegetation type within both segments. The high marsh of Seg. A was the only region where grain size changed with depth. In contrast in Seg. B communities, there were no difference in grain sizes with increasing depth.

Table 3.1 Average Percent Grain Size. Average percent grain size for each vegetation community within both segments for during 2010 and 2020. The number (n) of samples is the total number of samples for each vegetation community. Grain size percentages are an average across all depth intervals.

Year	Site	(n) Samples	Marsh Region	Dominant Vegetation	%sand	%silt	%clay
2010	Seg A	18	High marsh	*J. roemerianus*	65.34 ± 9.34	26.75 ± 8.42	7.95 ± 2.53
2020	Seg A	18	High marsh	*J. roemerianus*	58.75 ± 11.94	30.71 ± 9.71	10.55 ± 4.25
2010	Seg A	6	High marsh	*S. europaea*	69.33 ± 2.35	23.66 ± 2.93	7.01 ± 2.04
2020	Seg A	6	High marsh	*S. europaea*	61.70 ± 2.24	29.57 ± 3.45	8.73 ± 2.11
2010	Seg A	15	Mid marsh	*Mixed Meadow*	79.26 ± 4.19	16.98 ± 3.41	3.75 ± 0.99
2020	Seg A	15	Mid marsh	*Mixed Meadow*	71.60 ± 5.62	22.56 ± 3.25	5.83 ± 2.56
2010	Seg A	34	Low marsh	*S. alterniflora*	58.94 ± 6.06	33.30 ± 5.32	7.75 ± 1.48
2020	Seg A	33	Low marsh	*S. alterniflora*	54.26 ± 5.63	36.84 ± 4.76	8.90 ± 1.88
2010	Seg B	18	High marsh	*J. roemerianus*	75.62 ± 3.66	21.24 ± 3.20	3.14 ± 0.63
2020	Seg B	18	High marsh	*J. roemerianus*	62.48 ± 7.93	31.61 ± 6.18	5.91 ± 1.87
2010	Seg B	60	Low marsh	*S. alterniflora*	59.52 ± 6.08	33.59 ± 5.33	6.89 ± 1.50
2020	Seg B	60	Low marsh	*S. alterniflora*	50.47 ± 5.09	40.86 ± 3.97	8.68 ± 1.46

3.2. DRY BULK DENSITY

Dry bulk densities varied significantly across the marsh segments and ranged from 0.34 to 1.97 g cm^{-3}. Dry bulk density values significantly ($p < 0.0001$) increased over the decade within both segments. Seg. A increased from an average DBD of 1.08 ± 0.12 g cm^{-3} in 2010 to 1.22 ± 0.10 g cm^{-3} in 2020. Similarly, Seg. B increased from an average of 0.68± 0.09 g cm^{-3} in 2010 to an average of 0.74± 0.11 g cm^{-3} in 2020.

Dry bulk density was also compared across vegetation types and with increasing depth in samples collected in 2020. Seg. A had the greatest DBD values in the mixed meadow region (1.61 ± 0.10 g cm^{-3}) as a result of the higher %sand and minimal vegetation (Tables 3.1; Table 3.2). The other vegetation communities in the upper high marsh (*J. roemerianus*) and the lower marsh (*S. alterniflora*) had significantly ($p < 0.05$) lower DBD than the mixed meadow (Table 3.2). In Seg. B, trends were similar to those found in Seg. A, such that the upper high marsh (*J. roemerianus*) had a significantly ($p < 0.0001$) higher DBD than the low marsh (*S. alterniflora*) (Table 3.2). Overall, Seg. A (1.22 ± 0.10 g cm^{-3}) had significantly higher ($p < 0.0001$) average DBD than Seg. B (0.74 ± 0.11 g cm^{-3}).

Dry bulk density also varied with depth depending on the vegetation present. In Seg. A, the high marsh (*J. roemerianus*) and the mixed meadow were both characterized by a significant ($p < 0.05$) increase in DBD with increasing depth. Likewise, the DBD in Seg. B significantly ($p < 0.05$) increased with depth within the high (*J. roemerianus*) ($p = 0.02$) and low marshes (*S. alterniflora*) ($p = 0.009$).

Table 3.2 Average Dry Bulk Density, Percent Organic Matter and Nutrients. The average DBD, %OM, %OC and %TN and %TP for each vegetation community within both segments during 2010 and 2020. The number (n) of samples is the total number of samples for each vegetation community including depth sections. The values given are an average across all depth intervals.

Year	Site	(n) Samples	Marsh Region	Dominant Vegetation	DBD (g/cm^3)	%OM	%OC	%TN	%TP
2010	Seg A	54	High marsh	*J. roemerianus*	1.10 ± 0.13	8.88 ± 1.94	4.61 ± 1.42	-------------	-------------
2020	Seg A	54	High marsh	*J. roemerianus*	1.23 ± 0.14	9.70 ± 2.63	5.08 ± 1.59	0.23 ± 0.07	0.03 ± 0.01
2010	Seg A	18	High marsh	*S. europaea*	1.21 ± 0.08	6.67 ± 0.89	3.54 ± 0.80	-------------	-------------
2020	Seg A	18	High marsh	*S. europaea*	1.39 ± 0.07	6.15 ± 0.66	3.26 ± 0.57	0.16 ± 0.03	0.03 ± 0.00
2010	Seg A	45	Mid marsh	*Mixed Meadow*	1.34 ± 0.08	2.68 ± 0.94	1.10 ± 0.55	-------------	-------------
2020	Seg A	45	Mid marsh	*Mixed Meadow*	1.61 ± 0.10	2.40 ± 0.42	1.11 ± 0.27	0.06 ± 0.01	0.01 ± 0.00
2010	Seg A	99	Low marsh	*S. alterniflora*	0.91 ± 0.14	7.65 ± 1.32	2.17 ± 0.52	-------------	-------------
2020	Seg A	90	Low marsh	*S. alterniflora*	1.00 ± 0.09	7.14 ± 0.96	2.40 ± 0.46	0.18 ± 0.04	0.03 ± 0.01
2010	Seg B	54	High marsh	*J. roemerianus*	1.02 ± 0.13	6.21 ± 1.74	3.11 ± 1.19	-------------	-------------
2020	Seg B	54	High marsh	*J. roemerianus*	1.12 ± 0.12	6.69 ± 1.38	3.05 ± 0.70	0.14 ± 0.03	0.02 ± 0.00
2010	Seg B	180	Low marsh	*S. alterniflora*	0.57 ± 0.07	17.83 ± 2.46	8.51 ± 1.35	-------------	-------------
2020	Seg B	180	Low marsh	*S. alterniflora*	0.62 ± 0.10	17.84 ± 2.75	7.97 ± 1.42	0.38 ± 0.06	0.03 ± 0.00

3.3. ORGANIC MATTER

The %OM did not vary between 2010 and 2020, however it was characterized by significant spatial variability, with %OM ranging from 1.17 to 30.42%. The spatial trends in %OM were similar to those found in grain size and DBD. In Seg. A, high and low marsh %OM were similar and ~ 3 - 4 times higher ($p < 0.0005$) in %OM compared to the mixed meadow community (Table 3.2; Fig. 3.1). In Seg. B, the low marsh was a factor of two higher in %OM than the high marsh ($p < 0.0001$) (Table 3.2; Fig. 3.1). In comparisons across segments, only the low marsh was significantly ($p < 0.0001$) different, with the %OM in Seg. B more than double that measured in Seg. A (Table 3.2; Fig. 3.1). In general, %OM overall decreased with increasing depth, but was only significant ($p = 0.0002$) in the Seg. A high marsh region.

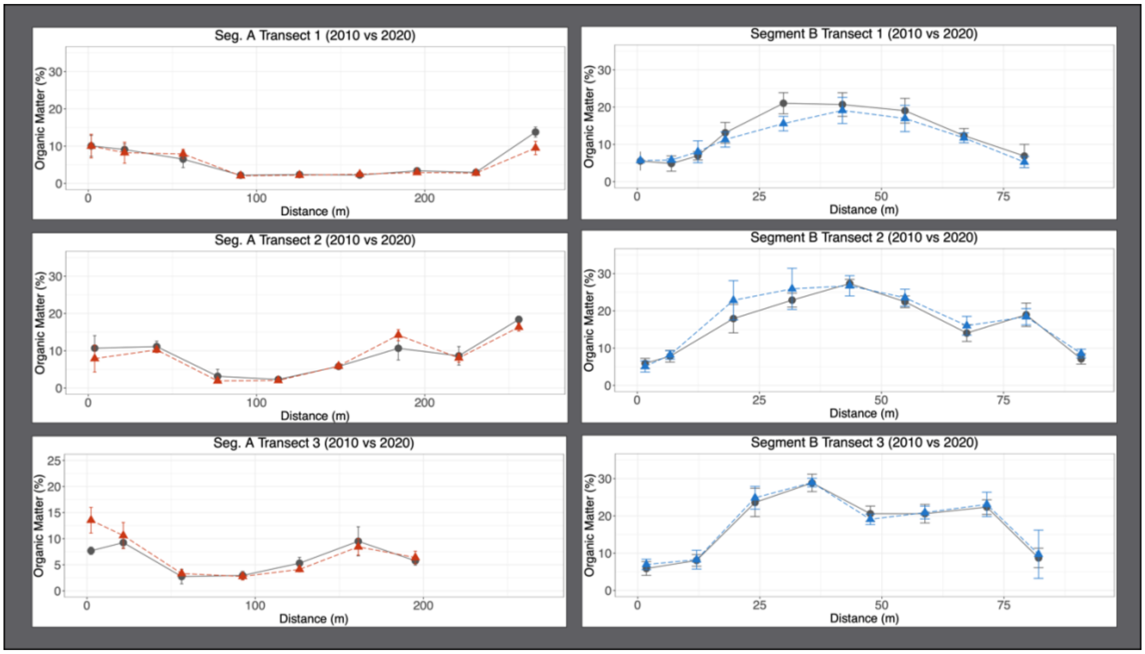

Figure 3.1. Average Percent Organic Matter of Plots. Seg. A (left) and Seg. B (right) displays the average % OM ($n = 3$) at each sample site with increasing distance from the forest edge to the creekbank for the years 2010 and 2020. Gray circles denote 2010 and triangles denote 2020. Red triangles denote Seg. A and blue triangles denote Seg. B.

3.4. CARBON, NITROGEN, AND PHOSPHORUS

PERCENT CARBON, NITROGEN AND PHOSPHORUS

Percent OC ranged from 0.25 to 13.91% and 1.33 to 14.61% in Segs. A and B, respectively, and showed no temporal trends, with spatial trends similar to that of %OM (Table 3.2). The %OC was highest in the low marsh (*S. alterniflora*) of Seg. B, which was again more than double that measured in the low marsh of Seg. A and across the high marshes in both segments. Percent TN showed similar trends to those of %OC, while %TP was not significantly different across segments or vegetation types (Table 3.2).

CARBON, NITROGEN AND PHOSPHORUS DENSITIES

Organic carbon, TN and TP densities were calculated using (Eq. 2) where $X_{density}$ is the density in g cm^{-3} for the element of interest, DBD is the dry bulk density in g cm^{-3} and %X is the percentage of a specific element of interest (e.g., %OC) determined from elemental analyses.

Equation (2)

$$X_{density} = DBD * \frac{\% X}{100}$$

Carbon densities showed significant differences across the marsh and ranged from 0.005 to 0.072 g cm^{-3}. Carbon densities in Seg. A significantly increased ($p < 0.0005$) from 2010 to 2020 from 0.022 ± 0.006 to 0.027 ± 0.006 g cm^{-3}, respectively. In contrast, Seg. B showed no significant temporal change. Nutrient densities were consistent with DBD trends.

Spatially, the highest C density measured between the two segments was found in the high marsh in Seg. A and in the low marsh in Seg. B (Table 3.3). The lowest C density was measured within the mixed meadow community and low marsh in Seg. A (Table 3.3). The high marsh was significantly higher ($p < 0.001$) than the low marsh for both segments and Seg. B had significantly ($p < 0.0001$) greater C densities relative to Seg. A. In general, C density showed no significant changes with depth except in Seg. A in the high marsh and low marsh ($p < 0.001$) and in Seg. B in the low marsh ($p < 0.0001$).

The average N density for Seg. A was significantly ($p < 0.0001$) lower (1.40 ± 0.32 mg cm^{-3}, range: 0.13 - 3.95 mg cm^{-3}) than Seg. B (1.88 ± 0.28 mg cm^{-3}, range: 1.03 - 3.15 mg cm^{-3}). N densities showed similar trends to C densities; the high marsh in Seg. A

and the low marsh in Seg. B contained the highest C densities, and the mixed meadow and the low marsh in Seg. A contained the lowest C densities (Table 3.3). In Seg. A, N densities were characterized by a general decrease with increasing depth, although this was only significant ($p < 0.05$) in the high and low marshes. Seg. B had no significant change in N density with depth.

Phosphorus densities showed no significant ($p < 0.39$) differences between segments (Table 3.3), with the highest P densities again found in the high marsh and lowest P densities measured in the mixed meadow. There were no significant trends in P density across Seg. B.

Table 3.3 Average Carbon and Nutrient Densities. Carbon and nutrient densities and C:N ratios for all sediment samples for this study during the year 2020 including all depth intervals. Densities and ratios are averaged for each vegetation type within both segments.

Year	Site	(n) Samples	Marsh region	Dominant Vegetation	C Density (g/cm^3)	N Density (mg/cm^3)	P Density (mg/cm^3)	C:N
2020	Seg A	54	High marsh	*J. roemerianus*	0.047 ± 0.011	1.99 ± 0.52	0.22 ± 0.04	29.6 ± 3.6
2020	Seg A	18	High marsh	*S. europaea*	0.037 ± 0.006	1.77 ± 0.30	0.22 ± 0.03	25.2 ± 3.2
2020	Seg A	45	Mid marsh	*Mixed Meadow*	0.017 ± 0.004	0.85 ± 0.20	0.11 ± 0.03	22.3 ± 4.2
2020	Seg A	99	Low marsh	*S. alterniflora*	0.018 ± 0.003	1.26 ± 0.26	0.18 ± 0.03	17.6 ± 2.0
2020	Seg B	54	High marsh	*J. roemerianus*	0.033 ± 0.007	1.52 ± 0.31	0.17 ± 0.04	25.9 ± 3.1
2020	Seg B	180	Low marsh	*S. alterniflora*	0.042 ± 0.006	1.99 ± 0.27	0.15 ± 0.02	24.4 ± 2.0

3.5. COMPONENT RELATIONSHIPS

ORGANIC CARBON VERSUS ORGANIC MATTER

Percent OM and %OC had strong correlations across vegetation communities and marsh regimes (Fig. 3.2). Relationships between %OM and %OC were examined using several different models identified in previous work (e.g., Craft et al., 1991). Here, a linear model best described the relationship between %OC and %OM ($R^2 = 0.93$) across all segments, including all 3 depth intervals, and was not significantly different from a

quadratic fit or other linear and quadratic relationship developed in prior studies in other regions (Fig. 3.2; Table 3.4). Linear regression models were further explored across vegetation types and only in Seg. A for *S. alterniflora*, the species that dominated the low marsh, had a significantly ($p < 0.0001$) lower slope (%OC = 0.28(OM) + 0.37) relative to the other vegetation types (green circles in Fig. 3.2).

Table 3.4 Percent Organic Matter and Organic Carbon Model Comparisons. Percent OM and %OC model comparisons from this study and other published work.

Reference	Region	Linear Model	R^2	Quadratic Model	R^2
This study*	North Inlet Estuary, SC	0.456(OM) – 0.135	0.93	$0.0023(OM)^2 + 0.3911(OM) + 0.1458$	0.93
St. Laurent, 2020	Delaware Bay Estuary, DE	0.415(OM) – 0.565	0.89	$0.002(OM)^2 + 0.293(OM) + 0.820$	0.90
Craft et al., 1991	Eastern North Carolina	---------------------	-----	$0.0025(OM)^2 + 0.40(OM)$	0.99
Chumra, 2003	Global	---------------------	-----	$0.0025(OM)^2 + 0.04(OM)$	-----
Holmquist, 2017	Conterminous United States	---------------------	-----	$0.074(OM)^2 + 0.0421(OM) – 0.0080$	0.93
Callaway, 2012	San Francisco Bay Estuary, CA	---------------------	-----	$0.001217(OM)^2 + 0.3839(OM)$	0.99

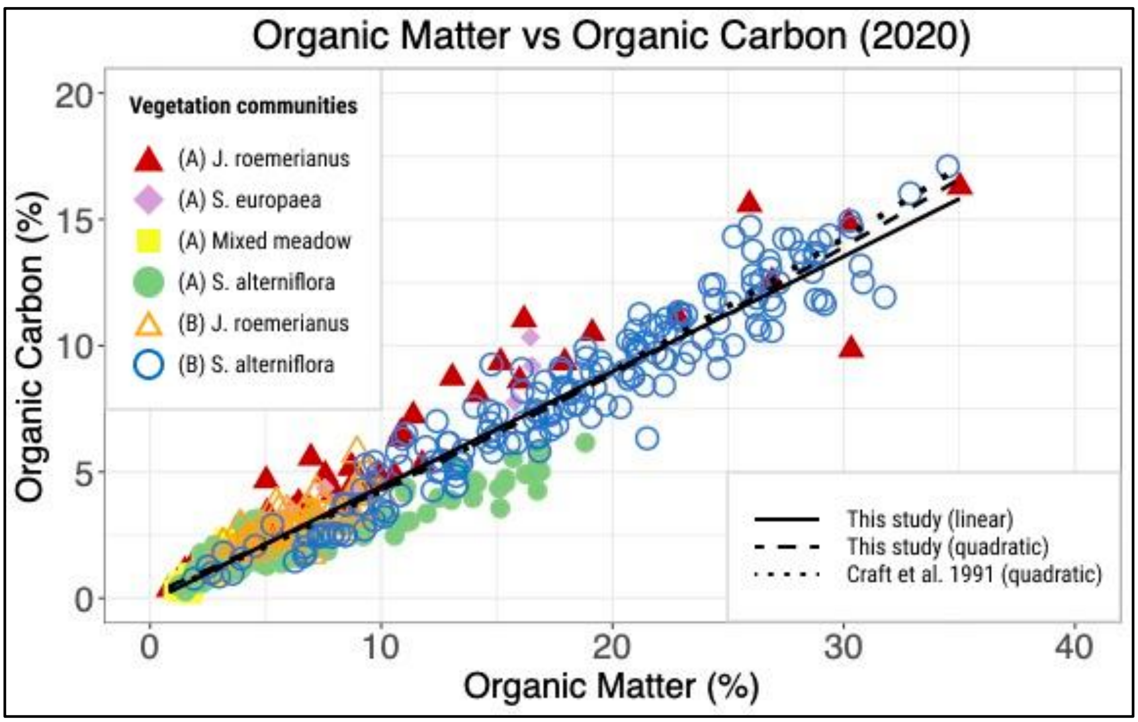

Figure 3.2 Percent Organic Matter and Percent Organic Carbon Model. Linear model estimates of %OC as a function of %OM for samples collected in the year 2020. Seg, A vegetation is denoted by the solid symbols: *J. roemerianus* (red triangles), *S. europaea* (pink diamonds), Mixed meadow (yellow circles) and S. alterniflora (green circles). Seg. B vegetation is denoted by the open symbols: *J. roemerianus* (orange triangles), and S. alterniflora (blue circles). This study's linear fit is shown (solid line) and the more commonly used quadratic fit (dashed line). The quadratic fit relationship from Craft et al., (1991) is shown as a dotted line for comparison.

DRY BULK DENSITY VERSUS ORGANIC MATTER AND ORGANIC CARBON

Relationships between DBD, %OM and %OC were examined and compared to previous work. In this study, a natural logarithmic model provided the strongest fit between DBD and %OM ($R^2 = 0.86$) and DBD and %OC ($R^2 = 0.72$) (Fig. 3.3; Table 3.5). Specific regions and vegetation types were further examined. Again, the low marsh *S. alterniflora* species in Seg. A had a significantly ($p < 0.0001$) lower slope (DBD = -0.563ln(OC) + 1.365, $R^2 = 0.80$) when compared to the other communities (Fig. 3.3).

Table 3.5 Dry Bulk Density and Percent Organic Matter Model Comparisons. Best fit %OM and %OC versus DBD relationships determined for this study relative to previous work.

Reference	Region	Model Type	Function	R^2
This study*	North Inlet Estuary, SC	Natural Logarithmic	$-0.48\ln(OM) + 1.9577$	0.86
This study*	North Inlet Estuary, SC	Natural Logarithmic	$-0.411\ln(OC) + 1.4639$	0.72
This study*	North Inlet Estuary, SC	Power	$1.461(OC)^{-0.446}$	0.65
Morris, 2016	Conterminous United States	Power	$0.569(OM)^{-0.31} - 0.55$	0.77
Callaway, 2012	San Francisco Bay Estuary, CA	Logarithmic	$-0.6922\text{Log}_{10}(OM) + 0.4133$	0.80

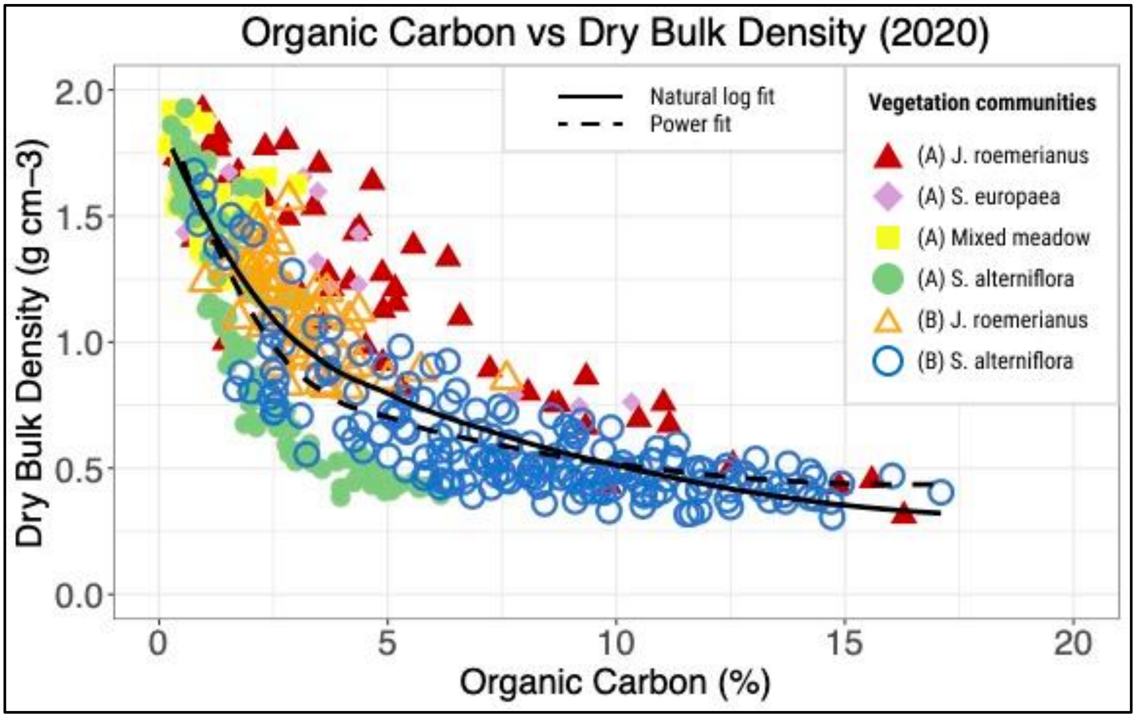

Figure 3.3 Dry Bulk Density and Percent Organic Carbon Model. Dry bulk density (DBD) versus %OC for all samples collected in the year 2020. Seg, A vegetation is denoted by the solid symbols: *J. roemerianus* (red triangles), *S. europaea* (pink diamonds), Mixed meadow (yellow circles) and S. alterniflora (green circles). Seg. B vegetation is denoted by the open symbols: *J. roemerianus* (orange triangles), and S. alterniflora (blue circles). Both a natural log fit derived from this study (solid line) and the power fit model are shown (dashed line).

CARBON VERSUS NITROGEN DENSITIES

Carbon and nitrogen density relationships were examined across specific regions of the marsh, vegetation communities, and with depth (Fig. 3.4). In general, there were no significant differences in the C/N density ratios expect for the Seg. A *S. alterniflora* community, which had a significantly lower C/N ratio with an average of 17.5 ± 2.0, (R^2 = 0.80) compared to the average of all other vegetation types 25.2 ± 2.8, (R^2 = 0.91) (Fig. 3.4).

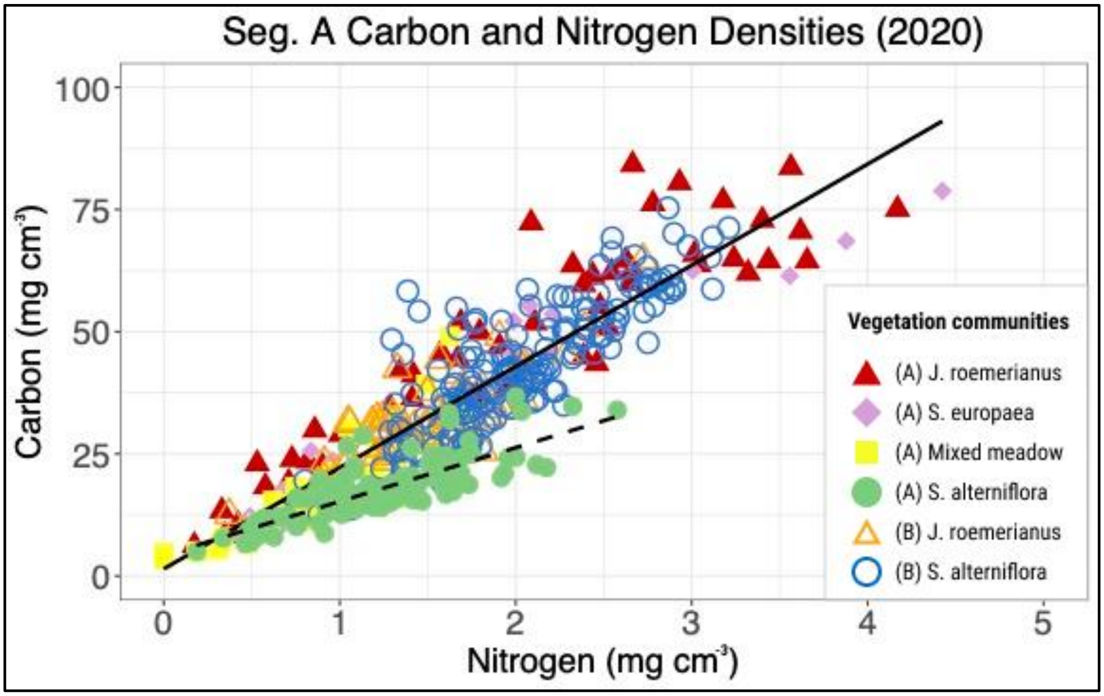

Figure 3.4 Carbon and Nitrogen Density Model. Carbon versus N densities (solid line). Seg, A vegetation is denoted by the solid symbols: *J. roemerianus* (red triangles), *S. europaea* (pink diamonds), Mixed meadow (yellow circles) and S. alterniflora (green circles). Seg. B vegetation is denoted by the open symbols: *J. roemerianus* (orange triangles), and S. alterniflora (blue circles). Only *S. alterniflora* in Seg. A had a significantly different slope from the other vegetation communities (dashed line).

CHAPTER 4: DISCUSSION

Saltmarshes store about 20-25% of the worlds soil organic carbon, 10 times more than their relative surface area (Duarte et al., 2013; Baustian, 2017). Carbon storage variability within saltmarsh ecosystems has been assessed by numerous studies in recent years, and there is rising interest in the economic benefits (e.g. carbon offsets, storm protection and food security) of blue carbon ecosystems, specifically in aiding climate change mitigation (Macreadie et al., 2019). Such studies have provided insight into the global and regional differences in carbon storage among several types of coastal ecosystems, including mangroves, seagrasses, salt marshes and brackish and freshwater wetlands. Fewer studies have focused on shorter-term decadal changes or on the small-scale spatial variability of carbon storage within a single tidal marsh. Here, we discuss similarities and differences in carbon and nutrient regional models of carbon storage and their importance to carbon stock assessments.

4.1. TEMPORAL VARIABILITY

Over the past decade, North Inlet has experienced a 20% increase in tidal inundation. While subtle changes in grain size have occurred, such as a reduction in sand and an increase in silt and clay that has increased DBD by 10-15%, vegetation communities, %OC, and carbon and nutrient densities have remained remarkably consistent. The decadal increase in silt and clay is consistent with the longer-term trend of suspended sediment importation of 0.55 kg s^{-1} into the North Inlet Estuary from the

coastal ocean and/or adjacent estuaries (Vogel, 1996). Current tidal gauge records at the South Carolina coast show a relative sea level rise of 3.4 mm y^{-1} while the North Inlet Estuary has a sediment accretion rate of ~ 2.72 mm y^{-1}, which so far has allowed the marsh to vertically accrete and migrate landward as tidal inundation increases (Vogel, 1996). Furthermore, the consistency of C to N density ratios across depth and therefore over time, apart from the *S. alterniflora* community in Seg. A, is likely a result of the active root zone and production occurring over the total depth measured. Combined, these, balanced processes over time and depth helps to explain the lack of decadal change in marsh composition.

Uncertainty remains as to whether the North Inlet marsh will remain stable over the next several decades as sea level continues to rise at a rapid rate. A theoretical model presented by Morris et al. (2002) suggests that there is an optimum rate of relative sea level rise at which the equilibrium elevation and depth of tidal flooding will lead to ideal conditions for primary production. However, if the equilibrium elevation falls below the optimum conditions for primary production, the marsh will fail to keep up with the acceleration of sea level rise.

This model suggests that this scenario will likely occur in most east coast estuaries when the local rate of sea level rise exceeds about 5 mm y^{-1}. The current rates of sea level rise along North Inlet and the east coast average 3.4 mm y^{-1} (Morris et al., 2016). and are close to the tipping point where long-term elevation loss and eventual drowning of coastal wetlands within the century will occur, and thus result in a loss of carbon storage potential (Morris et al., 2016).

The optimum rate of relative sea level rise will vary regionally due to differences in vegetation, tidal range and sediment and nutrient loading (Morris et al., 2016). For example, Louisiana's wetlands are currently experiencing a relative sea level rate of 9 mm y^{-1} due in part to rapid subsidence and reduced sediment supply that can lead to elevation loss, wetland drowning and the release of carbon stores back into the atmosphere (Arriola and Cable., 2017). (https://tidesandcurrents.noaa.gov/sltrends/sltrends.html).

4.2. SPATIAL VARIABILITY

Results confirm that while there are significant spatial differences in carbon storage within a single tidal creek, this variability can be constrained, in part, by relationships between DBD, %OM, %OC, and carbon and nutrient densities. These measured parameters allow for an improved understanding of the complex interplay of vegetation communities, geomorphology (age) and elevation gradients.

The range of %OM and %OC across all 2020 data was (0.8 to 35.0%) and (0.2 to 17.1%) respectively, with no consistent trends over elevation gradients or by vegetation communities. This study has shown that vegetation communities and marsh regions can display significant differences on the local scale. However, the range of variability seen across %OM, %OC and nutrients were reduced when accounting for the DBD.

Carbon densities ranged from (0.005 to 0.072 g cm^3) and instead displayed significant differences across elevation gradients between high and low marsh regions but no clear trend across segments. This could be a result of differences between geomorphology (age) of marsh segments and marsh structure.

When compared to other local east coast studies, we found a similar range in %OM within the Delaware Bay Estuary from (St. Laurent et al., 2020) whose freshwater wetland region ranged in %OM from (8.63 to 32.20%). The same study, however, found a higher range in %OM (6.87 to 52.97%) within a mesohaline marsh region. Both sites in the Delaware Bay Estuary contained the same *S. alterniflora* species as found in North Inlet. We found similar averages of %OM for *S. alterniflora* marsh region with Seg. A (7.14 ± 0.96%) and Seg. B (17.84 ± 2.75 %) compared to the Delaware Bay Estuary freshwater wetland (13.72 ± 4.05 %) and the mesohaline marsh (22.18 ± 10.69 %).

Another local scale study located near Jacksonville, FL from (Chambers et al., 2013) provided %OM and %OC by freshwater, brackish water, and salt marsh types. The vegetation species were similar, here they reported *J. roemerianus* in the brackish marsh and *S. alterniflora* in the salt marsh. Our average %OC including both segments averaged (4.93 ± 1.00%) with a range (0.19 to 17.10%) which was significantly lower than all other wetland types in their study, the low marsh *S. alterniflora* in Seg. B (7.91 ± 1.42%) was the only region closest to their values associated with a brackish marsh which had an average of (12.5 ± 1.1%).

A regional scale study conducted by (Callaway et al., 2012) within the San Francisco Bay Estuary, CA reported %OM and DBD measurements across brackish and saline marsh types. Compared to our study, they found opposite trends across the marsh platforms with %OM increasing and DBD decreasing across the marsh platform from low mash to high marsh. Their mean DBD values across brackish and saline marshes ranged from (0.21 to 0.65 g cm^3) while our DBD was ranged significantly higher (0.34 to

1.97 g cm^3). Their %OM was also significantly higher than our measurements across the brackish and salt marsh platforms.

We then compared results to another study from (Hansen and Nestlerode, 2014) who assessed several different wetland types on the regional scale spanning several states within the Gulf of Mexico. They state that their average soil OC densities ranged from (0.021to 0.090 g C cm^3) and there were no significant differences among different wetland classes, including freshwater marsh, brackish marsh, salt marsh, mangrove, and forested regions. Our total average C density (0.034 ± 0.006 g C cm^3) values across all data was not significantly different from the averages of salt marshes located in (0.035 ± 0.023 g C cm^3) Louisiana and outside (0.031 ± 0.023 g C cm^3) Louisiana according to this study. Additionally, they state there was no significant differences between the dominate vegetation communities of *J. roemerianus* and *S. alterniflora*, however, we found significant ($p < 0.001$) differences in C density between these two species within both segments. Furthermore, they reported the same *S. europaea* species in the Everglades National Park as having a high DBD (0.94 g cm^3) and low %OC (3%) but included another species *L. racemose* as being present. Our C density values for *S. europaea* however were significantly higher in DBD (1.39 ± 0.07 g C cm^3) but similar in %OC (3.26 ± 0.57 g C cm^3).

ORGANIC MATTER AND ORGANIC CARBON

There was a remarkably strong (R^2 = 0.93) linear relationship between %OM and %OC on the local scale within a salt marsh. Regional scale studies have modeled this relationship across multiple wetland types, from freshwater, to brackish to salt marsh systems, and have found a quadratic fit to best describe the relationship of %OM and

%OC. However, we found no significant differences between a linear or a quadratic fit within our study or across other work (Table 3.4).

We recognize that a linear fit may better describe a saline marsh type at a lower range of %OM (< 40%) when compared to the studies (Table 3.4) that included multiple wetland types and a higher range in %OM in their model. The high %OM range is typically found within many fresh and brackish marsh systems that may contribute to a better quadratic fit. For example, St. Laurent (2020) found a similar linear relationship between %OM and %OC likely due to their lower range in %OM, which was similar to that found in our study.

Overall, our assessment of our %OM and %OC linear model when grouped by vegetation communities, showed that the *S. alterniflora* community in Seg. A in the low marsh was the only region that showed significantly lower %OC relative to %OM. Although, we cannot determine the mechanism for this difference at this time, several factors could support this such as, organic matter degradation and changes in accumulation rates or biogeochemical processes.

We also confirmed that %OM and %TN showed a strong linear relationship with an ($R^2 = 0.92$) where %TN = 0.0212(OM) + 0.01, and a quadratic fit of ($R^2 = 0.92$), where %TN = $-0.0001(OM)^2 + 0.0244(OM) - 0.0039$. Few studies have reported this relationship, but we found that our quadratic model was not significantly different from Craft et al., (1991) relationship. Unlike our %OM and %OC model, there were no significant differences across all vegetation communities.

This study supports that loss on ignition (LOI) of soil organic matter is a reasonable proxy for estimating %OC and possibly %TN as an alternative approach to

elemental analysis (St. Laurent, 2020; Craft et al., 1991). Our models of %OM, %OC and %TN are only recommended for comparing with other estuarine marshes and mangroves, but it should be avoided when comparing with other wetland types until further evidence suggests this to be a reasonable conversion (Hansen and Nestlerode, 2014). These relationships will continue to improve as future studies will be carried out in North Inlet and across other estuarine wetlands. The establishment of these models will allow for more opportunities for sampling and analyses in coastal research. Loss on ignition is exceptionally useful for laboratories that lack the funding and instrumentation for elemental analyses (St. Laurent et al., 2020). The cost of sample processing for elemental analysis can range from $10 -$20 per sample whereas LOI requires simple equipment that reduces the overall cost of a project (St. Laurent et al., 2020).

DRY BULK DENSITY AND ORGANIC CARBON

Dry bulk density and OM, as described by Holmquist et al. (2017), are not independent variables but instead that DBD is a predictable function of %OM. As such, we found a strong (R^2 = 0.86) inverse relationship between DBD and %OM that showed a natural logarithmic fit for this small-scale study but some studies on the regional scale have found a power or logarithmic fit to be a better model (Table 3.5).

Few studies, however, have discussed model relationships between DBD and %OC. Here, we also found a close relationship (R^2 = 0.72) but more variability across certain vegetation communities when compared to our DBD and %OM model. Specifically, the *J. roemerianus* community in Seg. A showed a deviation (although not significant) of higher %OC from the general model (Fig. 3.3). We suggest that this community's variance may largely be driven by the differences in grain size compared to

the other high marsh communities. The *J. roemerianus* species in Seg. A contains nearly twice the amount of percent clay (10.55%) when compared to the *J. roemerianus* community in Seg. B (5.91%) and this region also contains the highest %clay across all vegetation communities (Table 3.1). The higher %OC in this community and segment may be due to the process of organic molecules that bind onto clay minerals more so than onto silt or sand sediments (Morris et al., 2016). This mechanism can raise the carbon content without adding to the total volume (Morris et al., 2016).

Additionally, the *S. alterniflora* community in Seg. A was significantly offset from the natural logarithmic model while all other vegetation communities were not significantly different. The largest deviation for this *S. alterniflora* community is prominent in the surface depth interval, but as there is an increase in DBD with increasing depth, this *S. alterniflora* community shows a closer fit to the general model. There was no significant difference in grain size with depth for either *S. alterniflora* community across segments that could otherwise suggest this variability. In summary, our model of DBD and %OC relationship has shown to be valuable for improving our understanding of carbon storage and variability and is suggested to be considered in future studies.

CARBON AND NUTRIENT DENSITIES

Carbon to nitrogen density ratios did not show significant differences across vegetation communities, segments, or depth with the exception again for the *S. alterniflora* community in Seg. A. Here, this community showed the lowest carbon to nitrogen density ratio with majority of this spread occurring in the surface depth interval. Few studies have reported this relationship, but we compared our ratios to those reported

in (Craft et al., 1991). Our *S. alterniflora* community in Seg. A had a ratio of (17. 5 ± 2.0) when compared to the average across all other vegetation types (25.2 ± 2.8). Craft described that a C:N ratio < 20 denotes a net mineralization of organic N while C:N ratio between 20 – 30, represent an equilibrium between mineralization and immobilization (Stevenson, 1986). Meaning that C storage potential may further be affected by biogeochemical processes and nutrient cycling.

C density values were calculated to assess C storage variability that is of further importance to measure C stocks. Our findings align with (Holmquist et al., 2017), who states that C density variability is not a well predicted by vegetation type. Instead, we found significant spatial variability between segments and between high versus low marsh regions. Such that, high marsh species within a given segment were not significantly different from each other and we see the same pattern for low marsh species. However, this trend was not consistent between Seg. A and B suggesting that geomorphology (age) and marsh platform structure, may play a role in estimating carbon storage.

When compared to the Delaware Bay Estuary study St. Laurent et al. (2020) they found significant differences in the average C densities across a mesohaline (0.054 ± 0.022 g cm^{-3}) and oligohaline (0.033 ± 0.001 g cm^{-3}) marsh regions. Within our study, Seg. A had an average C density of (0.027± 0.006 g cm^{-3}) while Seg. B had a greater C density of (0.040± 0.006 g cm^{-3}). The average C density value of Crab Haul Creek including both segments combined was (0.034± 0.006 g cm^{-3}), which fall within the range of the global average of (0.039 ± 0.003 g cm^{-3}) reported by Chumra et al. (2003).

This finding is an important consideration for sample collection because one should not collect sediments in only one region of the marsh or by species, which would result in one over or under estimating C stocks. We recommend instead considering the age and structure of the marsh platform when measuring C stocks.

CHAPTER 5: CONCLUSION

Research throughout the past decade has improved estimates of estuarine carbon storage dynamics at the regional scale (Macreadie et al., 2019). However, further interest is still growing in the area of small-scale spatial variability and the factors that control carbon and nutrient storage. This study contributes to the inventory of site-specific carbon storage values and to the relationships that govern sediment carbon variability. The two segments at North Inlet Estuary showed significant differences in carbon density values within a single tidal creek, where the younger marsh segment stored significantly more carbon than the older marsh segment. Vegetation communities did not show consistent spatial patterns and therefore were not useful for predicting carbon storage. There was no significant change in carbon storage over the decade, despite significant increases in inundation and rapid rates of sea level rise. Currently, there remains knowledge-gaps that limit the soil organic carbon and ecosystem change models (Spivak, 2019). Future studies need to collectively work towards standardized field and laboratory methodologies such as adopting the Blue Carbon Initiative (Spivak, 2019). Measurements of site-specific carbon storage will improve our current understanding of the role of local tidal marshes in carbon emissions and provide data to establish soil carbon models (St. Laurent et al., 2020). Providing a baseline and range of carbon density variability is therefore necessary to further inform coastal management decisions in restoration and preservation projects and to reach carbon emission reduction goals (St. Laurent et al., 2020).

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