Estuarine submerged aquatic vegetation habitat provides organic carbon storage across a shifting landscape

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HIGHLIGHTS

- Blue carbon estimates fail to include estuarine submerged aquatic vegetation habitats.
- Stocks of organic carbon across the estuarine submerged aquatic vegetation gradient were quantified.
- Fresh to saline estuarine SAV contains organic C stocks equivalent to seagrass, saltmarsh and mangrove.
- Estuarine SAV habitat retains significant blue carbon stock, and may offset emergent marsh loss.

GRAPHICAL ABSTRACT
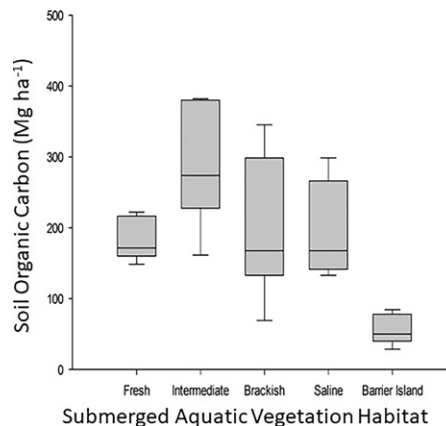
ABSTRACT

Submerged aquatic vegetation (SAV) thrives across the estuarine salinity gradient providing valuable ecosystem services. Within the saline portion of estuaries, seagrass areas are frequently cited as hotspots for their role in capturing and retaining organic carbon (C_{org}). Non-seagrass SAV, located in the fresh to brackish estuarine areas, may also retain significant soil C_{org}, yet their role remains unquantified. Given rapidly occurring landscape and salinity changes due to human and natural disturbances, landscape level carbon pool estimates from estuarine SAV habitat blue carbon estimates are needed. We assessed C_{org} stocks in SAV habitat soils from estuarine freshwater to saline habitats (interior deltaic) to saline barrier islands (Chandeleur Island) within the Mississippi River Delta Plain (MRDP), Louisiana, USA. SAV habitats contain C_{org} stocks equivalent to those reported for other estuarine vegetation types (seagrass, salt marsh, mangrove). Interior deltaic SAV C_{org} stocks (231.6 ± 19.5 Mg C_{org} ha^{-1}) were similar across the salinity gradient, and significantly higher than at barrier island sites (56.6 ± 10.4 Mg C_{org} ha^{-1}). Within the MRDP, shallow water SAV habitat covers up to an estimated 28,000 ha, indicating that soil C_{org} storage is potentially 6.4 ± 0.1 Tg representing an unaccounted C_{org} pool. Extrapolated across Louisiana, and the Gulf of Mexico, this represents a major unaccounted pool of soil C_{org}. As marshes continue to erode, the

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ability of coastal SAV habitat to offset some of the lost carbon sequestration may be valuable. Our estimates of C_{org} sequestration rates indicated that conversion of eroding marsh to potential SAV habitat may help to offset the reduction of C_{org} sequestration rates. Across Louisiana, we estimated SAV to offset this loss by as much as 79,000 Mg C yr^{-1} between the 1960s and 2000s.

1. Introduction

Although ecosystems store significant amounts of organic carbon (C_{org}) in plant biomass and organic-rich soils, there are currently few estimates in coastal regions (Keith et al., 2009; Mcleod et al., 2011). C_{org} storage occurs when atmospheric carbon dioxide (CO_2) is taken up by vegetation through photosynthesis, and is subsequently stored in plant biomass and soils, providing potential short-term (plant biomass) and long-term (soils) C_{org} storage. In coastal ecosystems, C_{org} is called blue carbon, with recent global stock estimates ranging from 140 to 471 Mg C ha^{-1} (CEC, 2016), and total sequestration rates exceeding 110 Tg C_{org} yr^{-1} (e.g., Mcleod et al., 2011; Duarte et al., 2013; Arias-Ortiz et al., 2018). Estimates of blue carbon stocks however remain focused on salt marsh, mangrove, and seagrass habitats (i.e., Mcleod et al., 2011; Ouyang and Lee, 2014; Thorhaug et al., 2019), with recent assessments expanding to new habitats (i.e., Macreadie et al., 2019). While these habitats capture blue carbon within the saline portions of estuaries, extensive coastal areas from fresh to saline marsh and submerged aquatic vegetation (SAV) may also provide similar storage capacity, especially in subtropical and tropical latitudes. In particular, across the extensive coastal areas of river deltas, blue carbon habitats occur across a wide range of tidal and salinity regimes. Given the dynamic nature of the estuarine landscape, which is constantly shifting due to subsidence, vertical accretion, sea-level rise, riverine inflow, erosion, habitat change, restoration and land building, accounting for blue carbon stocks would be more complete if it included estimates of C_{org} across the shallow-water estuarine gradient including SAV from freshwater to marine species. The inclusion of SAV C_{org} estimates in delta regions, extending from freshwater to marine habitats will provide a more accurate regional and global accounting of blue carbon stocks (Thorhaug et al., 2019).

SAV habitats share similar environmental settings and conditions conducive to soil C_{org} storage as other blue carbon habitats (i.e., Lavery et al., 2013; Kuwae et al., 2015). Specifically, high allochthonous and autochthonous organic matter deposition and low decomposition rates contribute to high amounts of C_{org} accumulation (sequestration) (Chmura et al., 2003; Mitra et al., 2005; Hopkinson et al., 2012; Kuwae et al., 2015). These conditions occur across the estuarine salinity gradient, which includes areas occupied by SAV. Further, the production and structure of estuarine vegetation, including SAV, provide autochthonous organic matter and restrict water movement, contributing to increased sediment deposition and potential C_{org} burial (Koch, 2001; Hendriks et al., 2008; Kennedy et al., 2010; Hillmann et al., 2019). SAV habitats across the estuarine gradient occupy similar geomorphological settings, which control soil C_{org} storage through local accretion rates, organic matter export and re-deposition rates (Bernal and Mitsch, 2012; Lavery et al., 2013; Smith and Osterman, 2014; Shields et al., 2017). For example, within seagrass meadows in Australia, high variation in C_{org} stocks was attributed to differences in species composition and environmental setting (i.e., inundation regimes, tide amplitude, sediment deposition environment; Lavery et al., 2013). Similarly, environmental setting resulting from the interaction of rivers, tides, waves, and climate modulates local accretion rates, organic matter export and re-deposition rates, helping explain variations in global carbon storage in mangrove wetlands (Twilley et al., 2018). If environmental and geomorphological drivers modulate vegetation production and C_{org} stocks

within coastal marsh and mangrove habitats, then SAV habitats, across the salinity gradient, may also contribute to capturing and retaining carbon.

Estuaries along the northern Gulf of Mexico extend from freshwater dominated estuarine depositional environments to saline barrier islands. These coastal regions have been identified as a blue carbon hotspot based on carbon stock estimates in mangroves, marshes, and seagrass habitats (Thorhaug et al., 2018; Thorhaug et al., 2019). Yet, these coastal areas also support extensive non-seagrass dominated SAV habitat across the fresh to saline coastal gradient, and within deltaic and barrier island settings (Hillmann et al., 2016). Recent work across the northern Gulf of Mexico estuaries estimated that 75% of this region exists as shallow aquatic open water habitat (Enwright et al., 2014). Of this shallow water habitat, an estimated 550,000 ha have been identified as potential SAV habitat (Hillmann et al., 2016; DeMarco, 2018) with an estimated 17% of this area covered by SAV (La Peyre et al., 2017; DeMarco, 2018). These estuarine environments are marked by rapid landscape changes including impacts from coastal erosion, subsidence, and river diversions, where these extensive shallow water areas represent potential SAV habitat. Thus, this habitat represents a transitional ecotone between emergent wetlands and mud-bottom habitats, providing a buffer to the effects of marsh loss, and supporting significant ecosystem services including long-term C_{org} storage.

Rapid changes in the estuarine landscape across coastal Louisiana include conversion of emergent marshes to shallow open water habitats, and shifting of estuarine marsh types (CPRA, 2017). Specifically, in the mid-1900s, there were an estimated 19,500 km^2 of coastal marshes, but by 2016 an estimated 25% of these marshes had converted to shallow open water (Couvillion et al., 2017). Further, of the remaining coastal marsh, loss rates differed by marsh types, which are defined by vegetation, and may differ in their biogeochemistry and carbon burial rates (i.e., Baustian et al., 2017). Specifically, intermediate marshes (i.e., dominated by *Spartina patens* with high species richness) increased by over 1000 km^2 while all other marsh types declined, with the largest declines in fresh and brackish marsh (>1000 km^2 each), and a smaller reduction in saline marshes (dominated by *Spartina alterniflora*; from data in Chabreck and Linscombe, 1982 and Sasser et al., 2014). Similarly, mangroves in coastal Louisiana have been reported since the early 1900s (Lloyd and Tracy, 1901) and are expanding, yet they are still not mapped separately from saline marsh and can't be accounted for separately in terms of carbon sequestration capacity.

Understanding how SAV habitat may contribute to carbon storage, and how changes in habitat extent and salinity may alter carbon stock and carbon sequestration rates, either in the short or long-term, remains a major knowledge gap when assessing blue carbon stocks. In particular, as we seek to fully define the extent of blue carbon habitats across an estuarine gradient, examining potential blue carbon habitats across a broad range of tidal and salinity impacts remains important. Identifying the full extent of a blue carbon habitat in terms of both marine influence, salinity and tidal effects is critical for (1) identifying if these habitats function equally in C sequestration, and (2) considering, in the face of landscape changes, how these habitats may be managed to enhance C stocks (i.e., Lovelock and Duarte, 2019). In this work, we quantified C_{org} storage across fresh to saline estuarine SAV habitats and estimated the potential effects of emergent marsh conversion to shallow water SAV habitat in the Louisiana delta plain. We hypothesized that SAV

habitat sequesters C_{org}, but that their contributions would vary in magnitude depending on the interaction between salinity gradients (fresh, intermediate, brackish, saline) and environmental setting (barrier island, estuarine) within coastal Louisiana, USA. Further, we hypothesized that C_{org} stored within shallow-water SAV habitats may partially offset the losses from emergent marsh erosion and loss in this region.

2. Methods

2.1. Study area

The Louisiana coastal plain is a dynamic environment, characterized by chenier plains in the west and by active and inactive delta lobes of the Mississippi River to the east (Fig. 1). Approximately 40% of the coastal marshes in the contiguous United States are in Louisiana and are surrounded by over 1.5 million ha of shallow ponds, lakes and bays (Chabreck, 1971). Soils in the region are characterized as histosols with high organic matter accumulation and sustaining high vegetation productivity. In southeastern coastal Louisiana, deltaic lobes of the Mississippi River Delta Plain (MRDP) consist of 1.2 million ha of coastal habitats delimited by Vermilion Bay to the west and the Chandeleur Islands to the southeast. Habitats within the MRDP are structured by salinity and hydrology (Visser et al., 2013) that are controlled by high interannual variation in seasonal riverine inputs (Mississippi River to the east, Atchafalaya River to the west; Bianchi and Mead, 2009). As a result, the deltaic region is a mosaic of diverse estuarine habitat types, ranging from forested swamps, to fresh through saline marshes, including, over 800,000 ha of extensive shallow-water (<2 m) habitats with fine sediment, oyster reefs, submerged and floating aquatic vegetation, and barrier islands. The Chandeleur Islands, a barrier island complex, formed by the reworked sediments of the abandoned St. Bernard Delta; the barrier islands are currently in a transgressive phase and strongly impacted by hurricanes and tropical storms (Bianchi and Mead, 2009; Fitzgerald et al., 2016).

Within this region, the coastal area is divided into marsh zones which are defined by vegetation alone but reflect effects of salinity and flooding on emergent marsh vegetation (Visser et al., 1998; Visser et al., 2013). These marsh zones define a gradient of salinity and hydrology across the area, and estimates of these areas and spatial layers are developed and updated regularly in Louisiana (i.e., CPRA, 2017). Additionally, recent work developed a spatial layer identifying SAV habitat for this region, providing spatial layers of SAV habitat, and estimates
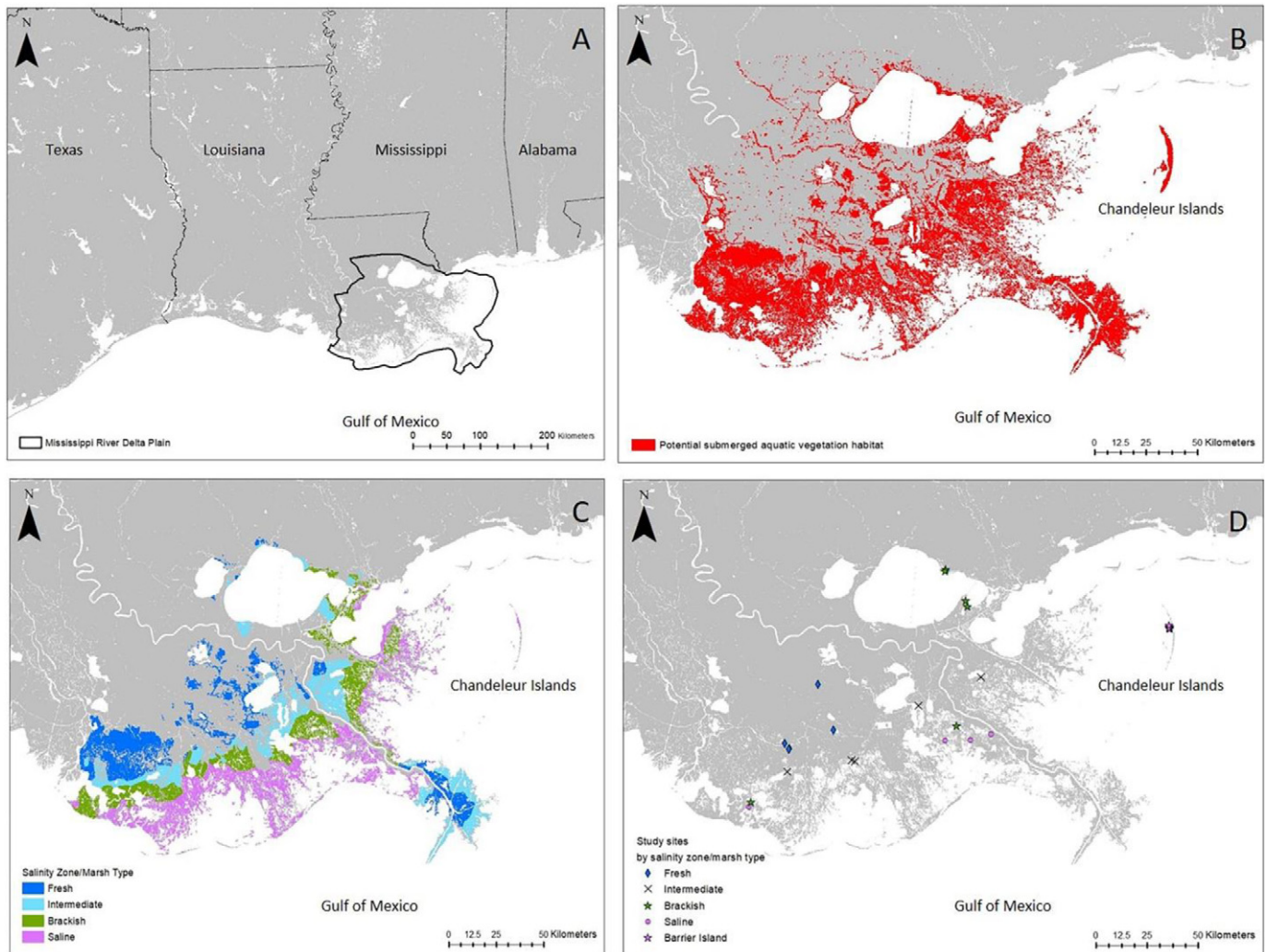

Fig. 1. (A) Map of study area within the Mississippi River Delta Plain. (B) Background map denotes potential submerged aquatic vegetation habitat area within the Mississippi River Delta Plain (140,000 ha). An estimated 20% (28,000 ha) are expected to contain submerged aquatic vegetation at any one time (La Peyre et al., 2017). (C) Map denotes marsh zone designations (Visser et al., 2013). (D) Site locations, distributed across submerged aquatic vegetation habitats (5 fresh, 5 intermediate, 6 brackish, 4 saline, 5 barrier island) across three Mississippi River basins (Terrebonne, Barataria, Pontchartrain) were sampled once in 2015 to 2016.

of percent presence of SAV within this layer (Hillmann et al., 2016). The SAV habitat layer identifies areas across the coast which, through the use of remote sensed and field data indicated SAV occurrence over the last 20 years. Methods to derive the spatial layers of SAV habitat incorporated the use of supervised classifications using remote sensed data from stacked Landsat satellite images (1984–2011), Landsat 8 remote sensing data, and selection of rooted vascular bed classes from National Land Cover Data (NLCD; Homer et al., 2012), and the Coastal Change Analysis Program (C-CAP; NOAA, 2012). The satellite classifications selected pixels that indicated SAV habitat in >10% of stacked images through either pre-defined SAV classes, or supervised classification of bands 3, 4, and 5 in the Landsat 8 remote sensing images. The NLCD and C-CAP datasets were used to select areas identified as rooted vascular bed classes that were identified through remotely sensed and field data (Hillmann et al., 2016). Using this approach, 140,000 ha were identified as SAV habitat within the MRDP (Fig. 1). An extensive 3-year field sampling across this region (2013–2015) and all salinity zones found that within this SAV habitat layer, 20% SAV cover was likely to be present (La Peyre et al., 2017). Thus, our current estimate of SAV coverage within the MRDP is 28,000 ha (140,000 ha * 0.2). Further work modeled the likelihood of SAV occurrence across this same region, identifying the percent likelihood of occurrence of SAV within each pixel. The modeling results revealed that SAV presence was predicted by winter mean salinity, turbidity and exposure (energy environment), with a correct classification rate of 74% (DeMarco et al., 2018).

2.2. Field sampling

Twenty-five sites were sampled in shallow water (<2 m) habitat throughout the MRDP (Fig. 1), representing four marsh types across the salinity gradient (i.e., deltaic fresh, intermediate, brackish, saline). Although these marsh types are defined by vegetation communities, they also reflect the regulatory effect of flooding and salinity regimes on emergent marsh vegetation (Visser et al., 1998; Visser et al., 2013). Since salinity is extremely variable within the MRDP due to seasonal river discharge, we used vegetation communities, representing long-term salinity and hydrology patterns, rather than discrete snapshots of salinity when selecting sites across the gradient. These sites were further located at sites known to support SAV (Hillmann et al., 2016; DeMarco et al., 2018). Additionally, twenty of the twenty-five sites were also co-located with Coastwide Reference Monitoring System (CRMS) stations that monitor hourly salinity, temperature °C, water level (CPRA, 2017). Saline habitats were further classified into two landscape categories: saline-estuarine and saline-barrier island sites. Barrier island sites were all established near the Chandeleur Islands. Due to logistical constraints, we selected 5 sites in each fresh, intermediate and saline barrier island habitat, 6 sites in brackish habitat and 4 sites in saline estuarine habitat. All sites were sampled once, either in summer 2015 or summer 2016.

Each site was located a minimum of 2-m from the marsh edge. Upon arrival at each site, water salinity (YSI-85, YSI Incorporated, OH), water temperature (°C, YSI-85, YSI Incorporated, OH), and turbidity (NTU-Hach 2100Q, Hach, CO) were sampled from an anchored boat. Afterwards, a 0.25 m × 0.25 m floating PVC quadrat was thrown haphazardly three times from the boat. Within each replicate, water depth was measured using a metric measuring stick (±0.01 m) and percent total SAV cover and cover by species were estimated using a modified Braun-Blanquet technique (Wikum and Shanholtzer, 1978).

Except for the barrier island sites, aboveground and below ground biomass samples were collected. Specifically, all aboveground (AG) biomass within each replicate quadrat was harvested at the sediment surface, placed in labeled plastic bags and placed on ice in a cooler. Belowground (BG) biomass was collected using a soil corer (10 cm interior diameter, i.d. x 30 cm length, L). Three BG cores were collected at each site with one core adjacent to each replicate AG quadrat. The BG core samples were rinsed in the field using a 1.0 mm mesh bottom

bucket to remove most of the sediment and reduce bulk before transfer to labeled plastic bags and placed on ice. All field samples were kept on ice and transported to the laboratory at Louisiana State University Agricultural Center (LSU) where they were stored at 4 °C until further processing.

Soil cores for carbon analysis were collected in duplicate and adjacent to each quadrat using PVC corers (5-cm i.d. x 100-cm L or 5-cm i.d. x 200-cm L, depending on site water depth). The corer was pushed evenly into the soil to collect a 1 m depth core, or until refusal; actual core depth penetration and extracted core length were recorded in the field. Any attached AG-SAV tissue was removed and extracted soil cores were then partitioned into 10 cm slices in the field, placed in individually labeled plastic bags, and transported on ice to LSU for processing.

2.3. Vegetation biomass processing

AG biomass samples were washed in the laboratory with tap water to remove sediment, debris, and epiphytes; then biomass was separated by species. Separated samples were dried at 60 °C to a constant weight (±0.001 g). BG biomass was washed over a 1.4 mm sieve to further remove sediment and debris and dried at 60 °C to a constant weight (± 0.001). AG and BG biomass samples from the barrier island SAV sites, where biomass samples were not collected, were estimated based on species-specific correlations between cover values and biomass from published values (*Ruppia maritima*; Hillmann et al., 2017, *Thalassia testudinum*: Congdon et al., 2017). AG and BG biomass was converted to g m^{-2}, and mean total SAV biomass per site was then converted to carbon equivalents based on published carbon content of SAV biomass (34.5%; live biomass carbon content (CC_{SAV}) = (AG + BG tissue biomass) * 0.35; Mg C ha^{-1}; Hillmann et al., 2017).

2.4. Soil analysis

Core sediment slices ($n = 330$) were dried at 60 °C to a constant weight (±0.001 g), and values used to calculate bulk density (BD) by dividing dry weight by volume of the slice and corrected for compaction and expansion (Howard et al., 2014). This correction factor was used across all sediment slices in a core. Soil compaction was <1% overall, but ranged between 10% for compaction and 10% expansion per core. After drying and weighing, sediment slices were homogenized using a mortar and pestle. To determine % dry weight organic matter (OM) per depth section, 4.0 g subsamples of each slice were weighed and then combusted at 550 °C for 4 h to calculate OM using the loss on ignition method (Heiri et al., 2001). To determine soil percent organic carbon (% DW, C_{soil}) per depth section, a set of subsamples were weighed out (0.030–0.040 g), placed in silver packages and analyzed in an elemental analyzer after HCl fumigation (Wetland Biogeochemistry Analytical Services Laboratory, Louisiana State University, Baton Rouge, LA; Zimmerman et al., 1997).

The organic carbon content (CC, g C g_{DW}^{-1}) of each sediment slice was calculated using the following equation (Campbell et al., 2015):

$$CC_A = Z_A * BD_A * (Csoil_A/100)$$

where A refers to the sediment slice (depth by site) and Z is the thickness of the individual slice (cm). To determine the C_{org} stock in the top 0–10 cm, 0–50 cm and 0–100 cm of soil, CC values were summed for each core and to the selected depth to provide total C_{org} stocks on an aerial basis. In shorter cores, linear and logistic regressions were used to estimate C_{org} values down core to 100 cm when appropriate (Fourqurean et al., 2012; Appendix A). When regression analysis indicated that down core trends in C_{org} values were not clear, we assumed a zero value for missing core slices (i.e., 21 out of 48 cores; or 44% of short cores). This procedure produced a conservative C_{org} stock estimate.

2.5. Statistical analyses

Discrete water column properties (salinity, temperature, water depth, turbidity), soil characteristics (BD, OM, $C_{soil,}$), vegetation biomass (AG and BG) and carbon content (C_{org}) were compared across SAV habitats using a generalized linear mixed model with a lognormal distribution and identity link function (Proc Glimmix; SAS, 2013). When significant differences were detected, both between and among groups F-statistic, degrees of freedom, and p-values were reported; post-hoc comparisons were estimated using the Tukey's procedure (SAS, 2013). Because the Spearman's correlation analyses showed a high correlation between C_{soil} and OM ($r > 0.93$; $p < 0.0001$), we only used C_{soil} in further analyses. Regression analysis was used to examine the relationship between SAV percent cover and C_{org} (100 cm; SAS, 2013). Mean and standard error are reported unless otherwise indicated; all tests are reported at a significance value of $\alpha = 0.05$.

2.6. Estimated coast-wide soil C_{org} sequestration rates

Using both published literature, and data from this study, we estimated soil C_{org} sequestration rates (Mg C y^{-1}) in coastal Louisiana emergent marsh and in shallow open water habitats likely to support SAV (hereinafter "SAV habitat"). In order to calculate these rates, we first estimated the rate of soil C_{org} sequestration by marsh zone within emergent marsh and SAV habitat separately (g C m^{-2} y^{-1}) using bulk density (g cm^{-3}), percent C_{org} and vertical accretion rates (mm y^{-1}) in fresh, intermediate, brackish and saline marsh and SAV habitat zones. Bulk density and percent C_{org} from Nyman et al. (1990) were used for determining marsh zone rates, while data from this study were used for SAV habitat zone rates.

Since published vertical accretion rates from the SAV habitat zones are not currently available, we used vertical accretion rates reported from MRDP emergent marshes (i.e., Nyman et al., 1990) for both the marsh and SAV habitat zones. The rates in fresh (6.7 mm yr^{-1}), intermediate (6.4 mm yr^{-1}), brackish (7.2 mm yr^{-1}), and saline (7.2 mm yr^{-1}) are similar to two published reports of vertical accretion in SAV in the Chesapeake Bay (Arnold et al., 2000; Palinka and Koch, 2012). Arnold et al. (2000) reported rates ranging from 5 to 7 mm yr^{-1} while Palinka and Koch (2012) found that vertical accretion rates were faster in persistent SAV beds (9 mm yr^{-1}) than in ephemeral SAV beds (3 mm yr^{-1}).

Using our calculated C_{org} sequestration rates by marsh and SAV habitat zones, we extrapolated Louisiana wide soil C_{org} storage rates by habitat type using the estimated area for each habitat type from the 1960s, and again for the 2000s. Estimated areas for the 1960s were taken from Chabreck (1970) for marsh zones, and Chabreck (1971) for SAV habitat (i.e., shallow water ponds in Chabreck, 1971). Only the area of shallow water ponds <260 ha was used for the calculations as Chabreck (1971) reported <0.05% SAV cover in the large ponds. The area for the small ponds was then multiplied by 24.8% to generate SAV habitat area for extrapolation as this was the estimated percent cover of SAV presented by Chabreck (1971). Estimated area for the 2000s for marsh zones was taken from Sasser et al. (2014). SAV habitat for the 2000s was estimated as the difference between emergent marsh area reported by Sasser et al. (2014), and the combined area of marsh and ponds reported by Chabreck (1970, 1971). This SAV habitat area was then divided into zones (fresh, intermediate, brackish, saline) assuming the same proportions as in Chabreck (1971). The use of two time periods enabled us to estimate changes in C_{org} sequestration over time, accounting for changes in the abundance of fresh, intermediate, brackish and saline emergent marsh and SAV habitat. These calculations use the best available data, and represent a first approach for estimating effects of changing estuarine environmental gradients, and habitat distribution on C_{soil} sequestration rates. We consider this methodology similar to the Tier I and Tier II level criteria as recommended by the IPCC (2006, 2013), and based on best available data when assessing wetland carbon sequestration and CO_2 emission rates from the local to regional to global spatial scales (IPCC, 2006, 2013, 2014; Villa and Bernal, 2018).

3. Results

3.1. Water column characteristics

Field water column salinity measurements differed significantly among SAV habitats ($F_{4, 20} = 10.73$, $p < 0.0001$) with higher values at the barrier sites (15.6 ± 0.1), lower values at the freshwater (0.1 ± 0.04), and no difference in mean salinity among intermediate, brackish and saline SAV habitats (Table 1). Turbidity ranged from 1.1 NTU to 89.9 NTU; there was a significant difference in turbidity among SAV habitats ($F_{4, 20} = 5.30$, $p = 0.0045$). Turbidity was lower at fresh and barrier island SAV habitats than in saline estuarine habitats, while brackish and intermediate habitats were similar to the other habitats. Water depth

Table 1

Soil and vegetation data by habitat type listed as mean (SE), and minimum-maximum range, including salinity, turbidity (NTU), water depth (cm), bulk density (g cm^{-3}), organic matter (%), C_{org} (%), C_{org10} (Mg ha^{-1}), C_{org50} (Mg ha^{-1}), C_{org100} (Mg ha^{-1}), SAV % cover, SAV aboveground (AG) biomass (g m^{-2}), SAV belowground (BG) biomass (g m^{-2}), CCsav AG (Mg C ha^{-1}), and CCsav BG (Mg C ha^{-1}). Habitat types are defined as deltaic fresh (F), deltaic intermediate (I), deltaic brackish (B), deltaic saline (S) and barrier island saline (BIS). AG and BG biomass at barrier island SAV sites, where biomass samples were not collected, were estimated based on species-specific correlations between cover values and biomass from previous field surveys (*Ruppia maritima*; $y = 2.0905x + 16.784$, $R^2 = 0.6$; La Peyre et al., 2017), or published values (*Thalassia testudinum*: $y = 1.14x + 80.40$, $R^2 = 0.77$; Congdon et al., 2017). Biomass was converted to g m^{-2} using a carbon content of SAV biomass of 34.5% (Hillmann et al., 2017).

SAV habitat		Fresh		Intermediate		Brackish		Saline		Barrier island	
		Mean (SE)	Min-max	Mean (SE)	Min-max	Mean (SE)	Min-max	Mean (SE)	Min-max	Mean (SE)	Min-max
Salinity		0.2 (0.04)	0–0.2	3.7 (1.7)	0.2–7.2	4.4 (1.0)	0.6–6.8	4.8 (1.8)	0.4–7.4	15.6 (0.1)	15.4–15.8
Turbidity	NTU	7.8 (5.1)	1.1–25.7	31.2 (17.0)	9–89.9	19.3 (8.5)	9.2–57.8	35.7 (10.7)	15.2–56.4	5.5 (1.56)	2.8–10.1
Water Depth	cm	83.6 (21.7)	10–125	43.3 (12.6)	20–73	61.8 (5.2)	45–75	45 (12.1)	24–70	80.6 (5.0)	72–93
Bulk Density	g cm^{-3}	0.3 (0.04)	0.2–0.4	0.2 (0.06)	0.1–0.4	0.4 (1.0)	0.1–0.8	0.4 (0.08)	0.2–0.5	1.5 (0.12)	1.2–1.9
OM	%	26.8 (3.8)	20.8–39.3	52.1 (6.8)	34.5–70.4	29.5 (8.8)	7.7–52.8	26.7 (4.9)	20–39.1	1.5 (0.3)	1.0–2.2
Corg	%	10.6 (2.4)	5.6–18.5	24.93 (3.1)	16–32.7	13.7 (5.0)	2.3–29	11.2 (2.2)	7.8–16.5	0.5 (0.1)	0.4–0.7
Corg - 10 cm	Mg C ha^{-1}	23.4 (4.6)	9–31.3	31.5 (1.8)	27.2–35.4	21.2 (5.5)	9.6–41.4	23.24 (6.2)	6.7–35.6	7.3 (1.0)	4.4–9.7
Corg - 50 cm	Mg C ha^{-1}	99.8 (11.0)	75.3–129.0	196.7 (19.8)	161.4–262.9	121.0 (24.0)	68.9–187.4	131.1 (12.5)	108.8–158.6	35.9 (2.6)	28.2–41.0
Corg - 100 cm	Mg C ha^{-1}	184.2 (16.2)	148.6–222.1	289.4 (46.8)	161.4–382.3	196.9 (46.7)	69–345.3	270.9 (33.1)	228.3–355.1	56.5 (11.6)	28.2–84.0
SAV_Cover	%	75 (18.5)	10–100	54 (11.5)	15–75	36.7 (16.3)	5–95	32.5 (6.9)	20–40	72 (15.9)	35–100
SAV_AG	g m^{-2}	141.7 (33.3)	27.3–198.9	151.1 (42.7)	25.7–256.6	26.6 (18.7)	0.8–110.0	27.4 (18.6)	0.8–74.2	162.5 (18.1)	120–194
SAV_BG	g m^{-2}	0.5 (0.5)	0–2.3	0.3 (0.2)	0–0.7	0.6 (0.3)	0–1.56	2.0 (1.7)	0.04–6.3	.	.
CCsav_AG	Mg C ha^{-1}	0.5 (0.1)	0.1–0.7	0.5 (0.1)	0.1–0.9	0.1 (0.06)	0–0.1	0.1 (0.06)	0–0.3	0.6 (0.06)	0.4–0.7
CCsav_BG	Mg C ha^{-1}	0.00	0–0.008	0.00	0–0.002	0.00	0–0.005	0.01	0–0.02	.	.

across all locations ranged from 10 cm to 125 cm, with no significant differences in mean water depth among SAV habitats. Temperature did not differ among SAV habitats.

3.2. Vegetation variables

3.2.1. Species composition

Nine SAV species were identified and species presence differed among habitats (Table 1). *Ceratophyllum demersum* and *Hydrilla verticillata* dominated freshwater communities, and were found at 4 out of 5 freshwater sites. Six SAV species were identified in intermediate and brackish SAV habitat, but community composition differed. *C. demersum*, *Myriophyllum spicatum*, *Najas guadalupensis* and *R. maritima* were frequently present at intermediate sites, while the dominant species at brackish sites were limited to *M. spicatum* and *R. maritima*. *R. maritima* was also most dominant in both the saline estuarine and barrier island sites. *T. testudinum* was observed at 3 out of 5 barrier island sites. The most frequently encountered species was *R. maritima* (12 out of 25 sites).

3.2.2. SAV cover and CC_{SAV}

Total vegetation biomass across all sites ranged from 0.07 to over 300 g m^{-2} and provide cover estimated from 5 to 100%. Biomass and vegetation cover were highly correlated ($r = 0.81$, $p < 0.001$; Table 1). CC_{SAV} ranged from 0.004 to 3.41 Mg C ha^{-1} and differed among habitats ($F_{4, 20} = 5.02$, $p = 0.0058$). Barrier island CC_{SAV} was significantly greater than brackish CC_{SAV}, but neither barrier island nor brackish CC_{SAV} differed statistically from fresh, intermediate or saline estuarine CC_{SAV} (Table 1).

3.3. Soil variables

Total core lengths ranged from 31 to 100 cm (Appendix A). Average core length in fresh, intermediate, brackish and saline habitat was ~60 cm; while in the barrier island habitat, average core length was ~40 cm. Because the statistical analyses showed that C_{org} and OM were positively correlated ($C_{org} = 0.4935*OM-1.2903$; $R^2 = 0.90$;

$p < 0.001$), further analysis was performed using only C_{org} and bulk density (BD). BD ranged from 0.07 to 1.76 gdw cm^{-3}, and differed among habitats ($F_{4, 20} = 14.11$, $p < 0.0001$) with the highest value in the barrier islands (1.5 cm^{-3} $\pm$ 0.05) (Appendix B). BD was variable with depth (down core), with no clear differences among SAV habitats (Appendix B).

C_{soil} differed across SAV habitats ($F_{4, 20} = 28.80$, $p < 0.0001$) (Fig. 2A), ranging from below detection (<0.02%) to 41.4%. Intermediate SAV habitat had the highest mean C_{soil} (24.3 $\pm$ 3.0%) which was statistically similar to fresh and brackish SAV habitats and greater than saline estuarine and saline barrier island habitat. In contrast, the barrier island SAV habitat C_{soil} was substantially lower (0.5 $\pm$ 0.1%) than all other SAV habitats. C_{soil} was variable with depth across all SAV habitats with no clear trends (Fig. 2B).

C_{org} stock differed significantly among habitats ($F_{4,20} = 13.76$, $p < 0.0001$); barrier island soils had less C_{org} stock than all other habitats, which did not differ from one another (Fig. 3). This pattern was consistent regardless of core length (i.e., 10, 50, 100 cm). There was no significant relationship between the SAV percent cover and C_{org} stock (Fig. 4).

3.4. Estimated coast-wide soil C_{org} sequestration rates

Across fresh to brackish marsh zones, SAV habitat areas were estimated to sequester C_{org} at faster rates compared to emergent marsh while C_{org} sequestration rates in saline zones were similar between emergent marsh and SAV habitat (Table 2). The faster sequestration rates in SAV habitat compared to emergent marsh followed patterns in soil BD more than patterns in % C (Table 2). Estimated C_{org} sequestration rates generally increased from fresh to saline habitats in emergent marsh zones (Table 2). Those patterns also appeared to reflect patterns in soil BD more than patterns in % C. In SAV habitat however, estimated C_{org} sequestration rates appeared slower at both ends of the estuarine gradient than in the middle of the estuarine gradient (Table 2).

Using these habitat specific C_{org} sequestration rates to extrapolate across coastal Louisiana resulted in an estimated sequestration rate of 2,738,000 Mg C y^{-1} in the 1960s, but by the early 2000s, conversion
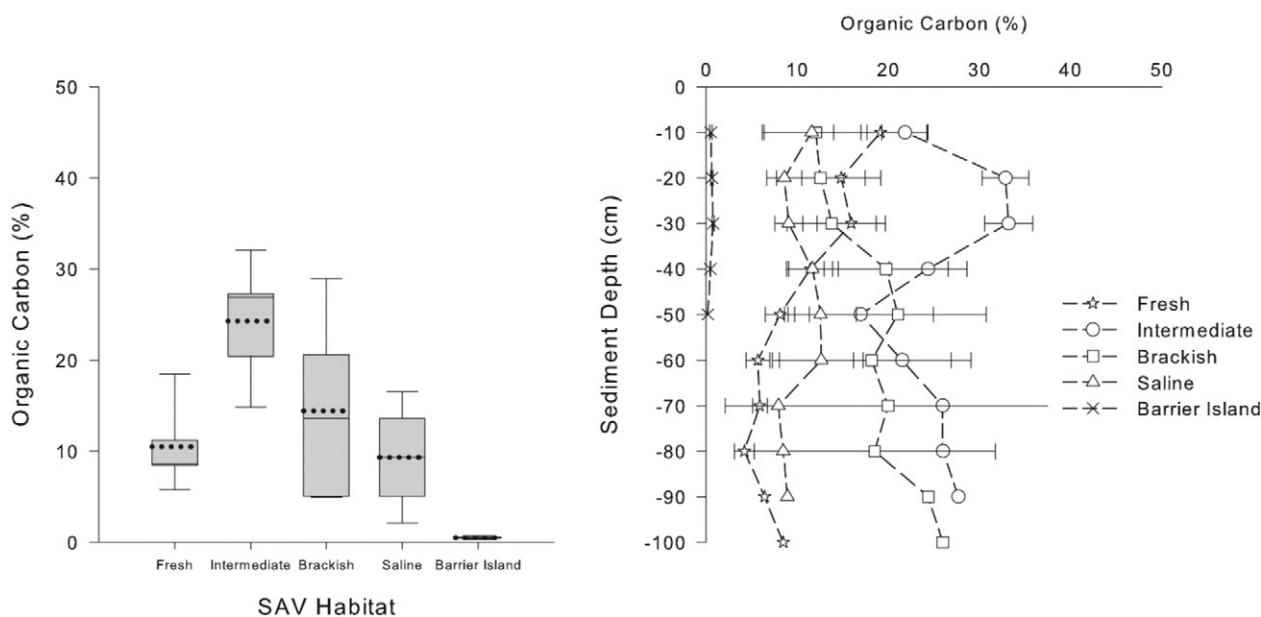

Fig. 2. Organic carbon (means ± 1 SE) across (left panel) submerged aquatic vegetation (SAV) habitats. Different letters indicate statistical differences. The dotted lines indicate mean values, with solid lines representing median values. Down core profiles of organic carbon (mean ± 1 SE) (right panel). SAV habitat is defined by marsh zones determined by long-term salinity (Visser et al., 2013) as fresh, intermediate, brackish and saline interior deltaic habitat, and saline barrier island habitat.
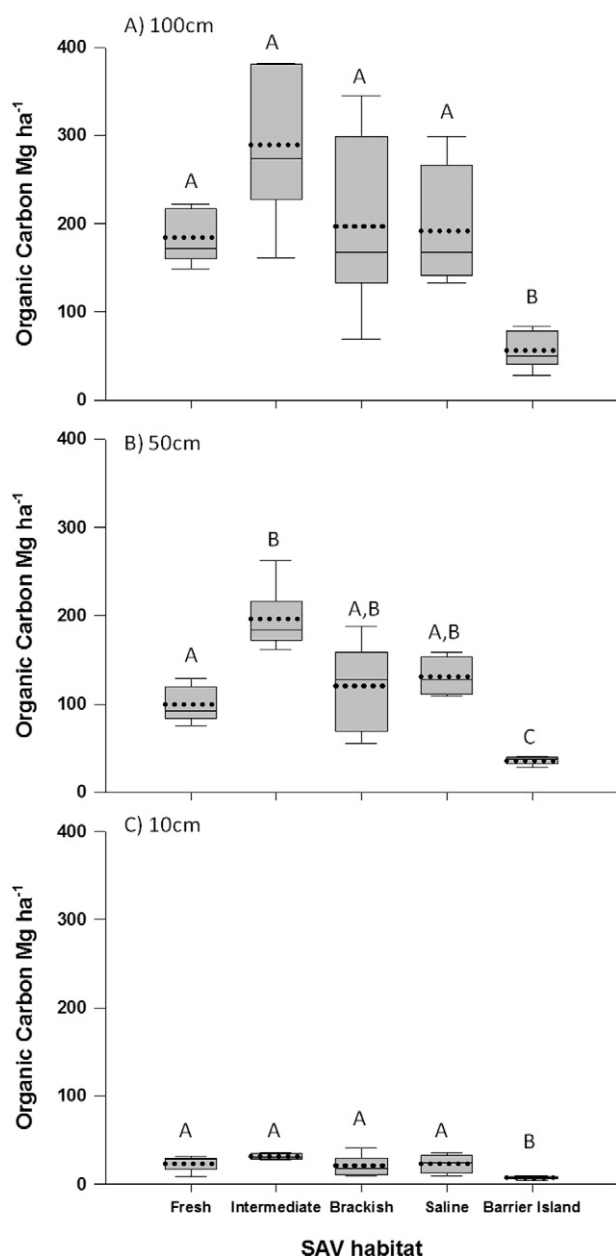
Fig. 3. Box plots of organic carbon stock (Mg C ha^{-1}) in top 100 cm (4A) of soil, top 50 cm (4B) of soil and top 10 cm (4C) of soil. SAV habitat is defined by marsh zones determined by long-term salinity (Visser et al., 2013) as fresh, intermediate, brackish and saline interior deltaic habitat, and saline barrier island habitat. Different letters indicate differences in means across habitats. Dotted lines indicate mean values. Black lines indicate median values.

Fig. 4. Relationship between submerged aquatic vegetation (SAV) cover (%) and soil organic carbon stock across salinity and settings for SAV. SAV cover (%) and biomass (g m^{-2}) were highly correlated. SAV habitat is defined by marsh zones determined by long-term salinity (Visser et al., 2013) as fresh, intermediate, brackish and saline interior deltaic habitat, and saline barrier island habitat. Symbols represent SAV habitat: fresh deltaic = ☆, intermediate deltaic = O, brackish deltaic = □, saline deltaic = △, barrier island saline = X.

ponds in the 2000s had changed in the same proportion as the emergent marshes (i.e., −5% fresh, +10% intermediate, −3% brackish, and − 2% saline), such that the relative abundances in the 2000s were actually 0.29 (fresh), 0.20 (intermediate), 0.41 (brackish) and 0.10 (saline), then the coastwide C_{org} sequestration would have been predicted to be 2,452,000 Mg C yr^{-1}. This then results in an estimated slowing of C_{org} sequestration by only ~286,000 Mg C y^{-1}, of which 365,000 Mg C y^{-1} is from a loss of wetlands, while conversion to SAV habitat offset this through a gain of an estimated 79,000 Mg C y^{-1}. In either case, without accounting for C_{org} sequestration in SAV habitat, coastal change in Louisiana would have been estimated to reduce C_{org} sequestration by ~365,000 Mg C yr^{-1}; accounting for C_{org} sequestration in SAV habitat changes those estimates to a reduction ranging by 75,000 to 79,000 Mg C yr^{-1} depending upon the distribution of marsh ponds across the estuarine gradient. These first-order estimates highlight the utility and significance of estuarine-wide assessment of the area and distribution of shallow water estuarine habitat across estuarine gradients, and the value of considering habitat conversions driven by environmental and geomorphological changes. Overlooking these historical changes in habitat distribution can significantly bias blue carbon sequestration rates.

4. Discussion

Submerged aquatic vegetation habitat C_{org} stocks across the MRDP were similar to stocks reported for other blue carbon ecosystems (i.e., 140–471 Mg C ha^{-1}; 34.9–509 Mg C ha^{-1}) (CEC, 2016; Thorhaug et al., 2018). When these values are extrapolated across the MRDP fresh to saline marsh zones using a mean SAV C_{org} stock of 231.6 ± 19.5 Mg C_{org} ha^{-1}, and an estimated SAV habitat of 28,000 ha, this region contains up to an estimated 6.4 ± 0.1 Tg C_{org}. In comparison, recent seagrass habitat estimates across Louisiana report a total of 0.2 Tg C_{org} (Thorhaug et al., 2019), suggesting a significant under-estimation of total SAV C_{org} storage. This potential contribution to regional carbon budgets from shallow water SAV habitats is significant given that approximately 75% of the coastal estuarine area across the northern Gulf of Mexico is shallow aquatic open water habitat (Enwright et al., 2014). As marshes continue to erode, the ability of SAV habitat to offset some of the lost carbon sequestration may be valuable. Recent modeling of SAV likelihood of occurrence across coastal Louisiana, indicated that likelihood of occurrence increased with increasing mean winter salinity,

of emergent marsh to ponds combined with changes in the estuarine gradient resulted in coastal Louisiana sequestering 2,448,000 Mg C y^{-1} (Table 2). This suggests an estimated slowing of C_{org} sequestration by 290,000 Mg C y^{-1}, with a loss of 365,000 Mg C y^{-1} from loss of emergent marsh, but a net gain of an estimated 75,000 Mg C y^{-1} through inclusion of SAV habitat. This represents an offset of close to 20% provided when sequestration rates account for and incorporate conversion of habitat types across the estuary.

The relative abundance of fresh (0.34), intermediate (0.10), brackish (0.44) and saline (0.12) potential SAV habitat was assumed to remain the same from 1960s to the 2000s as this potential SAV habitat area increased coastwide (Table 2). However, if the relative abundance of
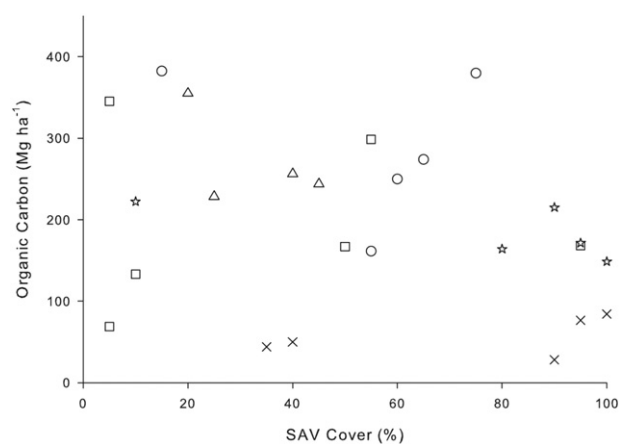
Table 2

Organic carbon storage rates in emergent wetlands and associated shallow open water habitats that are likely to support submersed aquatic vegetation across coastal Louisiana. Bulk density and percent organic carbon in shallow open water soil were estimated are from this study. Bulk density and percent organic carbon in emergent marsh are from Nyman et al. (1990). Vertical accretion rates reported from Nyman et al. (1990) for emergent marsh were used here for emergent marsh and for potential SAV habitat areas. The potential SAV habitat was multiplied by 24.8%, which is the mean percent SAV cover within those areas from Chabreck (1971).

Marsh type	Bulk density (g cm^{-3})	C_{org} (%)	Vertical accretion (mm y^{-1})	C_{org} rate per area (g C m^{-2} y^{-1})	Area, 1960s[a] (km^2)	C_{org} SAV storage rate, 1960s (Mg C y^{-1})	Area, 2000s[b] (km^2)	C_{org} SAV storage rate, 2000s (Mg C y^{-1})
Potential SAV habitat								
Fresh	0.28	10	6.7	187.6	2440	40,640	3242	55,778
Intermediate	0.20	24	6.4	307.2	742	20,203	984	27,729
Brackish	0.44	14	7.2	443.5	3169	124,752	4206	171,729
Saline	0.31	9	7.2	200.1	895	15,993	1191	21,951
Total potential SAV habitat					7246	201,589	9623	276,676
Emergent marsh								
Fresh	0.070	17	6.7	79.7	5258	419,259	3871	308,658
Intermediate	0.080	26	6.4	133.1	2777	369,721	3806	506,713
Brackish	0.160	16	7.2	184.3	5241	965,962	4036	744,004
Saline	0.240	12	7.2	207.4	3769	781,590	2954	612,535
Total emergent marsh					17,046	2,536,532	14,667	2,171,910
Total emergent marsh and shallow water					24,292	2,738,121	24,290	2,448,586

[a] Potential SAV habitat (shallow open water) area and marsh area during the 1960s was reported by Chabreck (1970, 1971). Potential SAV habitat is the reported area of small shallow open water ponds (Chabreck, 1971). The exact of year of the underlying map is unknown.

[b] Marsh area during the early 2000s was reported by Sasser et al. (2014). The exact of year of the underlying map is unknown. We assumed that the distribution of potential SAV habitat by zone were the same relative proportions as reported in Chabreck (1971).

and with decreasing turbidity and exposure (DeMarco et al., 2018). As Louisiana's coastal wetlands continue to erode, and anthropogenic activities influence riverine inflows, salinity, turbidity and exposure of shallow water areas will alter SAV maps; it is not clear what the balance will be between increasing versus decreasing salinity. For example, the change in salinity regimes may depend on a combination of factors including the extent of alteration to individual estuarine freshwater inflows, and the persistence of coastal marshes contributing to lowered turbidity and exposure. Given our current maps of SAV spatial extent, our estimates of C_{org} sequestration rates indicated that conversion of eroding marsh to potential SAV habitat may help to offset the reduction of C_{org} sequestration rates, with an estimated offset of up to 79,000 Mg C yr^{-1} across coastal Louisiana, between the 1960s and 2000s.

4.1. C_{org} stocks within estuarine SAV habitat

There were significant differences in C_{org} stocks between deltaic interior SAV habitats and barrier island SAV. Our study showed that the environmental setting likely plays an important role in differentiating between C_{org} stocks and sequestration rates in subtidal habitats, and, similar to other studies, that environmental factors such as geomorphology, allochthonous inputs, and hydrology can be used to improve predictions of blue carbon stocks, including estimates incorporating all SAV (Lavery et al., 2013; Ouyang and Lee, 2014; Gullström et al., 2017; Ricart et al., 2017; Twilley et al., 2018). The contrast between the larger soil C_{org} stock value estimated within interior deltaic habitats versus barrier island habitats may reflect longer water residence times, higher sedimentation rates, and less frequent erosional events than in the barrier island setting (Hedges and Keil, 1995; Röhr et al., 2016). For example, Röhr et al. (2016) found that *Zostera marina* thrives in Danish estuarine bays where autochthonous C_{org} is retained due to limited C export. In the MRDP interior deltaic SAV habitats, plants grow in protected waters that likely trap and retain SAV detritus and allochthonous organic matter from adjacent emergent marshes. Unfortunately, few studies have examined the fate and source of organic matter within this interior shallow water areas allowing proper quantification. This is similar for emergent marshes. Working in emergent marshes in the MRDP, Williams and Rosenheim (2015) concluded that C_{org} was more stable in fresh marsh than in brackish or salt marshes, yet similar data

from shallow water areas supporting SAV are lacking. In contrast to interior deltaic habitats, barrier islands are more frequently oxygenated by waves and occasionally shift location due to direct storm impacts, thus compounding the islands' transgressive movement towards the mainland. These physical drivers may affect not only SAV spatial distribution but also net primary productivity and soil organic matter decomposition significantly reducing organic matter deposition and storage.

Organic carbon retention also influences the observed differences in soil properties between the interior deltaic and barrier island sites due to the specific environmental setting drivers, regulating in situ organic matter production (DeLaune and White, 2012). For instance, soil C_{org} averaged <1% at the barrier islands but it was over 10% at the interior deltaic sites. Likewise, barrier island bulk density (BD: 1.5 ± 0.07 gdw cm^{-3}) was 5 times higher than interior deltaic BD (0.31 ± 0.02 gdw cm^{-3}) and similar to other studies reporting lower C_{org} and OM in seagrass bed soils (Duarte et al., 2013). Our BD values found within interior deltaic sites was similar to previously reported values, showing increasing BD with salinity in marsh habitats (Baustian et al., 2017; Wang et al., 2017). Those data were from surficial marsh wetland soils although we sampled subtidal areas. Furthermore, estuarine sites are generally characterized by fine clays and small particles while barrier islands are dominated by larger sediment grain size (e.g., sands); fine grained sediments have been shown to enhance Corg stocks (Dahl et al., 2016; Serrano et al., 2016).

Neither vegetation cover nor biomass was correlated with C_{org} stocks across the range of habitats sampled within the MRDP. This finding was consistent across a wide range of vegetation cover (5–100%) and biomass (0.5–300 g m^{-2}). In wetlands, vegetation influences C_{org} density by directly controlling net primary production, and indirectly, through enhancing sediment deposition (Gambi et al., 1990; Fonseca and Cahalan, 1992; Hendriks et al., 2008; Duarte et al., 2013; Armitage and Fourqurean, 2016). Thus, a decline in aboveground biomass from tidal fresh to saline zones (e.g. Hitch et al., 2011; Hillmann et al., 2019) suggest that blue carbon might also decline across estuarine gradients, however we did not detect such at trend in our study. This finding is similar to previous conclusions indicating that clear evidence of the quantitative relationships between vegetation cover or biomass and enhanced C soil stocks remains mixed, and potentially species-dependent (i.e., Lavery et al., 2013; Campbell et al., 2015; Dahl et al.,

2016). It also is possible that the lack of a negative decline in biomass along a salinity gradient is because soil organic matter across our study sites was dominated by organic matter that originated in adjacent emergent wetlands rather than from SAV in shallow water, indicating significant allochthonous inputs.

4.2. Soil C_{org} sequestration rates across an estuarine landscape

Additional research assessing the rate at which blue carbon pools might be changing due to natural and human disturbances is needed. In coastal areas where relative sea-level rise (i.e., the combination of global sea-level rise and local subsidence/uplift) is zero, soil organic matter content eventually can reach an equilibrium with a fairly stable carbon inventory despite on-going production, decomposition, and erosional processes, yet this mechanism is unlikely in our study area. This is because spatially variable subsidence (Nienhuis et al., 2017) creates a wide range of relative sea-level rise rates (Couvillion et al., 2017). Thus, the long-term blue carbon soil C_{org} sequestration rate should be estimated as the product of the vertical accretion rate (mm yr^{-1}), the bulk density (g cm^{-3}) and percent organic carbon. Although long-term vertical accretion estimates are available from emergent wetlands, we are unaware of vertical accretion rates obtained in adjacent areas supporting SAV. Erwin et al. (2006), using short-term measurements, observed that accretion rates in ponds generally exceeded accretion rates in adjacent emergent marsh. Thus, combining our BD and percent organic matter estimates with values of marsh vertical accretion in fresh, intermediate, brackish and saline marsh from Nyman et al. (1990) yielded first order estimates ranging from 187.6 g C m^{-2} (fresh ponds) to 443.5 g C m^{-2} (brackish ponds) (Table 2). Our SAV habitat blue carbon stocks were greater than those reported for adjacent emergent marsh, except for the saline marsh (Table 2); yet overall they were in agreement with previous estimates for the same habitats (Baustian et al., 2017). When these rates are extrapolated coastwide, SAV habitats potentially sequester 0.3 Tg C_{org} yr^{-1}, and marshes 2.2 Tg C_{org} yr^{-1}.

Natural (e.g., storms) and anthropogenic (e.g., induced land loss) disturbances can cause blue carbon to re-mineralize and release CO_2 back into the atmosphere or to enter adjacent coastal water as DOM, DIC and DOC (Pendleton et al., 2012; Hopkinson et al., 2012; Williams and Rosenheim, 2015). The fate of carbon from eroding marshes in the MRDP, including the impacts of large storm events, could affect carbon stocks, long-term carbon storage (Delaune and White, 2012; Williams and Rosenheim, 2015), and regional landscape level carbon budgets (Hinson et al., 2017). Interestingly, numerous studies have measured Corg in coastal wetlands and spatially extrapolated Corg across these habitats. However, previous estimations did not consider the effects of emergent wetlands converting to shallow water (SAV) habitats, and possible switches from fresh to saline-dominated environments (Hinson et al., 2017). Our estimates (Table 2) are the first attempting to evaluate the relative contribution of SAV that are common throughout the MRDP and Chenier Plain of coastal Louisiana. These estimates depend upon the spatial distribution of marsh ponds across the estuarine gradient, which has not been re-evaluated since 1968 (Chabreck, 1971). If we had only considered the decline of emergent wetland area, we would have determined that coastal change in Louisiana had slowed C_{org} sequestration by ~365,000 Mg C yr^{-1}. But, accounting for C_{org} sequestration in ponds, whose aerial extent across the estuarine gradient hasn't been measured since the 1960s (Chabreck and Linscombe, 1982), led to estimates indicating that coastal change, accounting for SAV, reduced the slowing of C_{org} sequestration by 75,000 to 79,000 Mg C yr^{-1} depending upon the distribution of marsh ponds

across the estuarine gradient in the 2000s. Despite our values' uncertainty, they highlight the utility and ecological significance of repeated estimates of open water areas supporting SAV across estuarine gradients, including the need to consider habitat conversion.

5. Conclusion

Updated and more robust estimates of blue carbon pools, including a wider range of in situ sequestration rates will contribute significantly to a better understanding of the global carbon budget in coastal areas. Specifically, these estimates will benefit most from direct measurements of accretion rates in SAV habitat, and the systematic improvement of mapping the distribution of shallow-water areas among fresh, intermediate, brackish and saline classes. Additionally, it is critical to evaluate the fate of soil organic carbon in emergent wetlands that shift to SAV habitat due to erosional processes or limited vertical accretion. Our estimates now explicitly include these processes for the first time by incorporating SAV habitats distributed across fresh to saline marshes in the MRDP. Our results also revealed that habitat change and emergent marsh annual loss reduce blue carbon storage capacity in emergent marshes by as much as 0.4 Tg C_{org}. While carbon stored in the MRDP may be lost through erosion (DeLaune and White, 2012), some of this carbon may be retained and preserved in shallow-water habitats soils (Wright and Nittrouer, 1995; Corbett et al., 2006; Allison et al., 2007). Understanding the functional interactions among accretion rates, bulk density, and carbon content regulating Corg stocks is critical to reduce major uncertainties in the construction of regional carbon budgets in the MRDP coastal region (Williams and Rosenheim, 2015; Ward et al., 2017).

Author contributions

ERH, MKL, VHR, JAN contributed to idea development. ERH collected data, processed samples and managed all daily activities. ERH, MKL analyzed data. ERH, MKL, VHR, JAN contributed to manuscript writing. Funding was provided through a grant to MKL, JAN.

Appendix A

Frequency count of complete cores versus cores with extrapolated data, A) 10 cm cores, B) 50 cm cores, and C) 100 cm cores. Frequency count represents the longer of two cores extracted per site. For all core size classes, black represents complete cores and white represents short cores. At 10 cm 100% of cores were complete; at 50 cm 86% of cores were complete; at 100 cm 4% of cores were complete. Regression analysis dictated extrapolation of C_{org} in short cores (fresh: $y = -1.5967x + 27.204$, $R^2 = 0.816$; intermediate: $y = -3.1705x + 46.377$, $R^2 = 0.6145$; brackish: $y = -1.7395x + 29.253$, $R^2 = 0.71$; saline: $y = 0.0441x + 25.936$, $R^2 = 0.2446$; barrier island: $y = -0.6188x + 9.058$, $R^2 = 0.6448$).
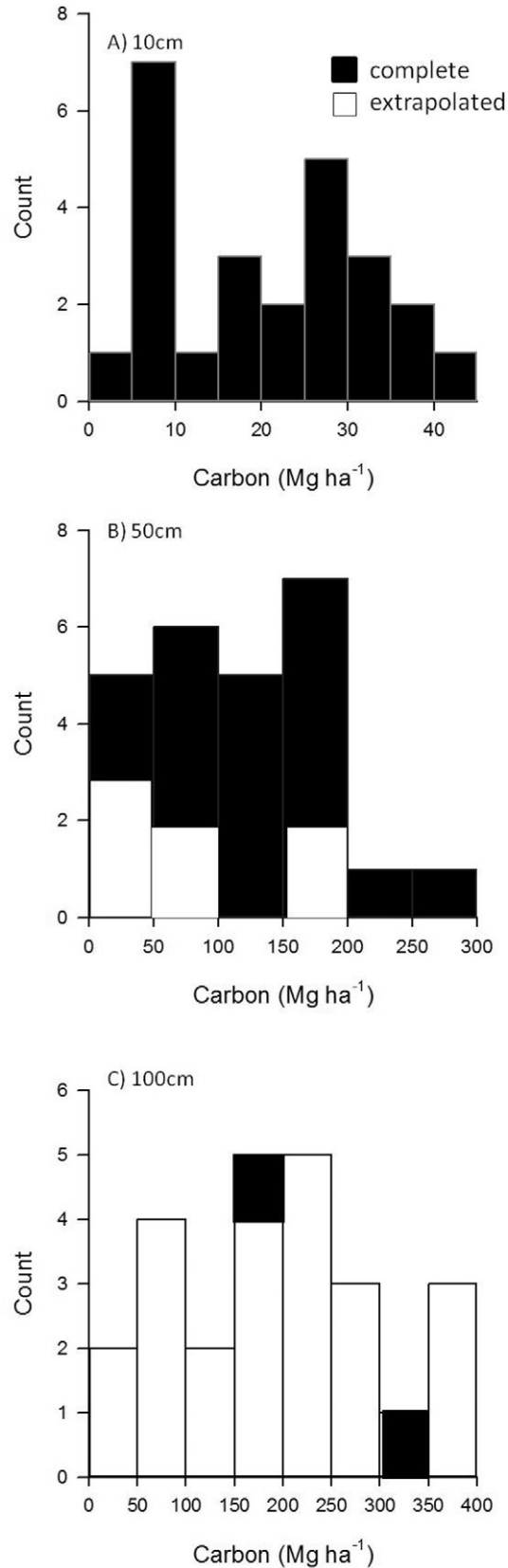
Appendix B

Bulk density (gdw cm^{-1}; mean $\pm$ 1 SE) across submerged aquatic vegetation (SAV) habitats within the Mississippi River Delta Plain (left panel). SAV habitat is defined by marsh zones determined by long-term salinity (Visser et al., 2013) as fresh, intermediate, brackish and saline interior deltaic habitat, and saline barrier island habitat. Different letters indicate statistical differences. The dotted lines indicate mean values, with solid lines representing median values. Down core profiles of soil bulk density (mean $\pm$ 1 SE) (right panel).

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