TIDAL WETLAND SOIL ORGANIC CARBON ACROSS THE CONTERMINOUS UNITED STATES

A Dissertation

by

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Submitted to the Office of Graduate and Professional Studies of
Texas A&M University
in partial fulfillment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

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December 2017

Major Subject: Ecosystem Science and Management

ABSTRACT

Tidal wetlands contain large reservoirs of carbon in their soils and sequester carbon dioxide (CO_2) at greater rates per unit area than nearly any ecosystem. The spatial distribution of this carbon influences climate and wetland policy. To assist with international accords such as the Paris Climate Agreement, national-level assessments such as the United States (U.S.) National Greenhouse Gas Inventory, and regional and local evaluation of CO_2 sequestration credits, I developed a geodatabase (CoBluCarb) and high-resolution maps of soil organic carbon (SOC) distribution by linking National Wetlands Inventory data with U.S. Soil Survey Geographic Database. For over 600,000 wetlands, total carbon stock and organic carbon density was calculated at 5-cm vertical resolution from 0 to 300 cm depth. There are 1,153-1,359 Tg of SOC in the upper 0-100 cm of soils across a total of 24,945.9 km^2 of tidal wetlands, twice as much carbon as the most recent national estimate. To assist conservation efforts and better understand the biogeochemical processes of these wetlands, I determined the statistical correlations of 45 different environmental variables and 5 different aspect factors with the distribution of this SOC. Environmental variables were divided into oceanic, terrestrial, and geographic variables to understand the array of potential influences. Geographic variables were the strongest predictors of SOC. Longitude correlated reasonably well with SOC density at the national scale (r^2 = 0.52), Gulf Coast (r^2 = 0.51), and West Coast (r^2 = 0.84). To determine SOC outward flux, I created current status maps showing wetland accretion, soil respiration, and remaining SOC stocks for the coastal wetlands in

the conterminous United States. The calculated outward fluxes and remaining stocks were (1) from soil to the atmosphere, (2) from soil to surrounding water, and (3) the remaining recalcitrant stock of SOC. Predictive maps estimated spatial distribution of the fluxes, providing a comprehensive overview in the coastal US. Overall, regional scales may provide the most promise for predicting SOC. It is possible to use standardized values at a range of 0-100 cm of the soil profile, to provide first-order quantification and to evaluate future changes in carbon stocks in response to environmental variables.

ACKNOWLEDGEMENTS

I would like to thank my committee chair, Dr. Rusty Feagin, and my committee members, Dr. Marian Eriksson, Dr. Thomas Boutton, and Dr. Anthony Filippi, for all their inputs and experiences that provided me with the knowledge and opportunities for following my passion.

Thanks to my friends and colleagues for all the motivation and support throughout my time at Texas A&M University.

Finally, thanks to my mother and father for their encouragement and support that allowed me to achieve my goals.

CONTRIBUTORS AND FUNDING SOURCES

Contributors

This work was supervised by a dissertation committee consisting of Dr. Rusty Feagin (advisor) and Dr. Marian Eriksson of the Department of Ecosystem Science and Management, Dr. Thomas Boutton of the Department of Ecosystem Science and Management, and Dr. Anthony Filippi of the Department of Geography.

All work for the dissertation was completed by the student, in collaboration with Dr. Rusty Feagin of the Department of Ecosystem Science and Management and of Dr. Marian Eriksson of the Department of Ecosystem Science and Management.

Funding Sources

Graduate study was supported by a fellowship from Texas A&M University and a dissertation research fellowship from the Tom Slick Foundation.

This work was made possible in part by NASA's Carbon Cycle and Ecosystems Program under Grant Number NNX14AM37G. Its contents are solely the responsibility of the author and do not necessarily represent the official views of the National Aeronautics and Space Administration.

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CHAPTER I

INTRODUCTION

Wetlands are a fundamental part of coastal ecosystems and influence the health of coastal communities. These ecosystems execute functions such as prevention of coastal erosion, filtration of estuarine waters, carbon sequestration, flood control, provision of a nursery habitat for marketable fisheries, and nutrient cycling into the oceans (Cline *et al.,* 2007). However, wetlands are now beginning to be seen as significant carbon storage areas with marketable value (Hopkinson *et al.,* 2012; Chmura 2011; Mcleod *et al.,* 2011). When analyzed on a per-acre basis, wetlands are the largest producer of ecosystem services per hectare of all the major terrestrial ecosystems, as a result of high biodiversity and variety of ecosystem services and watershed support (Brown and Lant 1999; Akin *et al.,* 2003; Mcleod *et al.*, 2011). Many times, wetlands are disregarded because many of the services cannot be correctly valued in economic terms and since wetlands are typically not aesthetically pleasing or easily used by the public, they are undervalued and seen as wastelands. The undervaluing of these resources results in conversion to other uses that create an immediate monetary benefit (Costanza *et al.,* 1997; Lui *et al.,* 2010; Brown and Lant 1999).

Before the 1970s, land conversion for agriculture made up the greatest portion of loss for wetlands, with an estimated annual net loss of about 455, 000 acres and an estimated 185, 700 acres lost per year for non-agricultural conversion (Brown and Lant 1999). From 2004 to 2009, there was an estimated 1.4% loss of coastal wetland area and an estimated 99 percent of all salt marsh decline was due to consequences from land subsidence, coastal storms, sea level rise, and ocean processes (Dahl, 2009). Indirect human influences related to climate change had the greatest impact and were the greatest cause of loss in the salt marsh wetlands, particularly in the Gulf of Mexico, which contains 40% of all the saltwater wetlands in the United States (Dahl, 2009; Edwards and Proffitt, 2003). Deforestation and land change across all ecosystem types have been determined to be the second leading factors for carbon dioxide into the atmosphere, releasing approximately 8-20% of all emissions (Pachauri and Reisinger, 2007; Bianchi *et al.*, 2012).

Since the time of the Industrial Revolution about two centuries ago, human inputs of greenhouse gases, namely carbon dioxide, have been altering the global atmosphere (Couto *et al.*, 2013). A warming and changing climate can drive sea level rise and alter other environmental factors such as the quantity of precipitation and mangrove encroachment, resulting in an indirect human influence on wetland loss (Couto *et al.*, 2013; Pachauri and Reisinger, 2007; Bianchi *et al.*, 2012). Even in the absence of complete loss, fragmentation of a wetland can cause basic processes to fail and a decrease in biodiversity, leading to a delicate and weakened environment (Cline *et al.*, 2007).

Coastal mangrove forests are able to accumulate large amounts of carbon and store it for many years due to their high rates of primary production and the relatively recalcitrant nature of woody plant tissues (Kristenson *et al.*, 2008; Fujimoto *et al.*, 1999; Matsui, 1998). Many times, the carbon stored will be resident for long periods of time, for example the soil carbon in a Brazil mangrove wetland was found to have a residence time ranging from 400-770 years in the upper 1.5 meters of the sediment profile (Kristenson *et al.*, 2008; Dittmar and Lara, 2001). For salt marshes, as compared with woody mangroves, there is relatively more carbon deposition belowground than in aboveground biomass (Chmura, 2011). Mangroves and salt marshes are two of the most productive portions of the estuarine and terrestrial systems, leading to large amounts of tidal wetland soil carbon or otherwise termed, blue carbon (Hopkinson *et al.*, 2012; Mcleod *et al.*, 2011).

CHAPTER II

SPATIAL VARIATIONS IN THE TIDAL WETLAND SOIL ORGANIC CARBON IN THE CONTERMINOUS UNITED STATES [1]

II. 1 Introduction

Tidal wetlands are among the most biologically productive and societally valuable ecosystems in the world (Costanza *et al.*, 1997; Martínez *et al.*, 2007; Barbier *et al.*, 2011), yet they continue to be lost at a global rate of approximately 1.5% annually (Hopkinson *et al*., 2012; Pendleton *et al*., 2012). The pace and scale of these losses has focused attention on the strategic need for initiatives that promote conservation and sustainable restoration of the physical landscape (Day *et al*., 2007). One strategy to sustain wetlands includes incentivizing public and private interests to begin accounting for "blue carbon," carbon sequestered by vegetated coastal ecosystems for long-term storage (Howard *et al.*, 2014).

Given the global extent of tidal wetlands and their high levels of productivity, blue carbon is a potentially active sequestration component for atmospheric carbon dioxide (CO_2) (Chmura *et al*., 2003; Laffoley & Grimsditch, 2009; Mcleod *et al*., 2011). Sequestration per unit area in these systems is estimated to be as much as 3 to 50 times greater than that of rainforests (Bridgham *et al.*, 2006; Nellemann *et al.*, 2009;

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Breithaupt *et al.,* 2012). The annual sequestration potential of blue carbon, not accounting for the current pace of coastal land loss, is estimated to be 0.9% to 2.6% of total anthropogenic CO_2 emissions (Murray *et al.*, 2011). Within the United States, coastal wetlands are attributed with 36% of the total sequestration by all wetlands and 18% of the total carbon sequestration of all ecosystems in the conterminous United States (Bridgham *et al.*, 2007).

A national scale accounting of this carbon resource has not yet been realized, though several U.S. agencies and institutions are currently engaged in the first nationwide inventory of coastal wetland carbon and GHG emissions, to be included in the annual Inventory of U.S. Greenhouse Gas Emissions and Sinks, and published by the United States (U.S.) Environmental Protection Agency (EPA). The EPA is applying new procedures provided by IPCC to recognize changes in carbon stocks associated with human activities (IPCC, 2014). A Tier 2 analysis (out of three) is being tested, whereby country-specific activity data and emissions factors are applied (Hiraishi *et al.*, 2014). A spatially explicit database that details blue carbon stock sizes and locations is needed for the first Tier 2 estimation of wetland soil organic carbon for the entire US.

The carbon in tidal wetlands also can have private economic value if managed for sale through offset transactions (Duarte *et al*., 2005; Crooks *et al*., 2010; Needelman *et al*., 2012; Wylie *et al*., 2016). Ecosystem management projects across the US created and sold approximately 30 million carbon credits to voluntary buyers in 2011, worth $180 million (Peters-Stanley *et al*., 2012). While coastal markets continue to expand (Grimsditch *et al*., 2013; Lau, 2012; Ullman *et al*., 2012), neither the spatial distribution of carbon nor the monetized credits themselves should be considered homogeneous (Marland *et al*., 2001; Miles & Kapos, 2008). An important piece of the puzzle is the ability to predict the geographic locations that offer the greatest potential for carbon management and profit (Crooks et al., 2009).

The overall objective was to delineate the geographic distribution of soil organic carbon (SOC) across the tidal wetlands of the continental US at the highest possible resolution. The abundance of SOC is largely determined by soil texture, climate, vegetation, and historical and current land use and management (Amundson, 2001; Howard *et al.*, 2014). I sought to compare SOC across a variety of categories, such as wetland types, US states, coastlines, and estuarine basins. In order to accomplish this task, I created the CoBluCarb database by combining National Wetlands Inventory (NWI) data with US Department of Agriculture (USDA) Soil Survey Geographical (SSURGO) data.

II.2 Materials and Methods

The creation of CoBluCarb involved three main steps. First, tidal wetland locations and types were extracted from the NWI. Second, measurements of organic matter fraction (OMF) and bulk density (BD) were extracted from SSURGO, and then used to compute the organic carbon density (OCD) and soil organic carbon stock, where possible, at 5-cm increments within individual SSURGO map units. Third, OCD and soil organic carbon stock were computed for individual wetland polygons by area-weighting map units within each wetland polygon. Once CoBluCarb was created, I evaluated its usefulness by comparing it to the literature. Finally, I mapped, summarized, and characterized the carbon distribution across various categorizations of tidal wetlands.

II.2.1 National Wetlands Inventory Dataset

Tidal wetland locations and types were extracted from the NWI database to create a dataset at a scale of 1: 24,000 that only included tidally influenced classes (Cowardin *et al.*, 1979; Federal Geographic Data Committee, 2013). The NWI uses a classification system for aquatic habitats that includes systems, subsystems, and classes. The boundary between the estuarine versus riverine and palustrine systems in the NWI data is where the salinity is equal to 0.5 parts per thousand during the period of annual average low flow. The NWI dataset variously defines freshwater tidal wetlands within palustrine (P) and riverine (R) categories, though without specific tidal subtype categorization except within the riverine category. Consequently, I created the specific requirement that “S”, “R”, “T”, or “V” modifiers had to be listed (each referring to a given NWI tidal regime for a tidal freshwater wetland: temporarily, seasonally, semi-permanently, or permanently flooded respectfully) in the palustrine wetlands to be considered tidal. Our dataset does not include any subtidal or supratidal subsystems, nor does it include the aquatic bed, reef, rocky shore, rock bottom, unconsolidated shore, unconsolidated bottom, and streambed classes within the estuarine intertidal and riverine tidal subsystems. It should be noted that other special modifiers, such as artificial, partly drained or ditched, farmed, etc. may be present throughout the tidal wetlands. ArcGIS was used to extract relevant wetland classes from the full NWI database, removing any spatial overlap of individual wetland polygons to avoid double-counting. I then combined the extracted files into four tidal wetland classes emergent vegetation (EM; largely equivalent to brackish to saline salt marsh), shrub-scrub (SS), forested (FO; largely mangroves), and freshwater tidal (FT; including herbaceous, shrub and forest vegetation).

II.2.2 USDA SSURGO Dataset

The USDA SSURGO database (Soil Survey Staff, 1993) was used to ascertain the OMF (mass organic matter per mass soil) and bulk density (mass of soil per unit volume at a water potential of 33 kPa) of a soil at all possible locations within tidal wetlands. In general, bulk density and organic matter fraction have a strong inverse relationship and organic matter and carbon content are also related, typically determined through loss on ignition method (Callaway *et al.*, 2012; Morris *et al.*, 2016). The SSURGO dataset is based on field coring data and interpretation from USDA Natural Resources Conservation Service soil scientists, with accuracy dependent on field and laboratory work (Zhong & Xu, 2011). To create accurate lab results, multiple samples from a given soil horizon were analyzed, with typically between one and three sites chosen for detailed analysis. Pits were dug to ensure the correct amount and profile of soil, in a stair-like fashion or straight-walled, typically 0.6 m × 2 m wide, with depth as required. Samples were taken from each horizon, and at times for sub-horizons. When horizons were included, a sufficient sample was required to create an accurate representation of the soil profile (Soil Survey Division Staff 1993; Zhong & Xu, 2011).

Soils of the same general type occur at several locations on the landscape; these are called *map units* (MUs). The SSURGO dataset is built on specific soil volumes that were sampled, rather than entire areas of these lands, with specialist interpretation of what SSURGO calls the soil *components* that comprise them. Thus, while relatively homogenous, the different MUs include one or more different components. Further, soils have depth and properties that vary by horizon. The number and depth of each horizon typically varies by component. For example, 239 separate wetland polygons in St. Bernard Parish, LA are associated with the MU 375349, which is named “Bellpass muck.” Bellpass muck has three separate components, with the Bellpass series comprising approximately 80% of the soil, and the Clovelly and the Lafitte series about 10% each. Unlike this example, the component percentages within a MU do not always sum to 100%, which is one source of “missingness” within the data that I describe in detail below. The Bellpass series has three horizons extending to depths of 65, 80, and 200 cm respectively. The Clovelly series has two horizons extending to depths of 71 and 200 cm and the Lafitte series has two horizons that extend to 190 and 200 cm. Bellpass muck is unusual in that all of its components extend to the same depth (200 cm). Careful accounting was made of these variations as well as others leading to several types of missingness identified in the CoBluCarb wetland carbon geodatabase. There were no other sources of soil property information considered beyond SSURGO in our further calculations, though the database could be adjusted or mined for future refinement, particularly as relates to adjustments on bulk density or hydric soil classification (such work is on-going as part of a related project, though I do not present this work here).

II.2.3 CoBluCarb

SSURGO provides information regarding the organic matter as a percentage, OM (%), which is the mass of organic matter over the mass of the soil by component and by depth in grams ($g_{om}\ g_{soil}^{-1}$ *100%), though I can easily transform this organic matter percentage into the organic matter fraction (OMF) by dividing by the total 100% ($g_{om}\ g_{soil}^{-1}$). Together with bulk density, BD ($g_{soil}\ cm^{-3}$) and the van Bemmelen constant, $v = 0.58$ *($g_{soc}\ g_{om}^{-1}$), a given cubic centimeter of soil (component *c* with bottom depth *b* within the *h*-th horizon's depth-range, $b \in [min_{hb}, max_{hb})$, gives the organic carbon density (OCD) in $g_{soc}cm^{-3}$ by

$$OCD_{cb(h)} = OMF_{cb(h)} \cdot BD_{cb(h)} \cdot v \qquad (1)$$

The above equation can be thought of as the density of SOC in the cubic centimeter at bottom-depth *b* below a 1 cm^2 area on the surface, if that cm^3 of soil is composed strictly of component *c*. The subscript *cb (h)* is used to emphasize the fact that a given cm^3 of soil (with bottom-depth *b*) is assumed to be within one and only one horizon. The validity of using the van Bemmelen constant for all tidal wetlands in the United States can be found documented in other studies (Zhong & Xu 2011; Keller *et al.,* 2015; Pribyl, 2010), but could provide fertile ground for future refinement.

Also for a single component only, the SOC to depth *d* is given in $g_{soc}cm^{-2}$ by

$$SOC_{cd} = \sum_{b=1}^{d}\sum_{h=1}^{n} OCD_{cb(h)} \cdot \iota_{cb(h)} \qquad (2)$$

where $\iota_{cb(h)}$ is an indicator taking on the value 1 if component *c* is present in the horizon *h* at the bottom depth *b* and 0 otherwise, and *n* is the number of horizons. Since the indices *b* and *h* both relate to soil depth, for any combination of *b* and *h*, at most one of the indicators will take on the value of 1. For example, for the Clovelly series in the Bellpass Muck MU above, horizon 1 is present at the depth of 40 cm, so $\iota_{Clovelly,40(1)}$=1 while $\iota_{Clovelly,40(2)} = \iota_{Clovelly,40(3)} = 0$. Similarly, $\iota_{Clovelly,90(2)} = 1$ while $\iota_{Clovelly,90(1)} = \iota_{Clovelly,90(3)} = 0$. One can create various depth bins (or, sub-portions of the vertical profile) by allowing *b* to begin at depths lower than below the bottom of the first centimeter.

Equations (1) and (2) consider that under a given square centimeter of area, the soil underneath it belongs to one and only one component. This situation is not always the case. Thus, a final summation operation allows components that are completely mixed within the MU for consideration. The percentage composition of a component as a proportion of the map unit was used as a weight, and a weighted average value was derived for organic matter fraction and bulk density, such that all components were used. In particular, the amount of SOC to depth *d* present under a square centimeter on the surface in $g_{soc}cm^{-2}$ was taken to be

$$SOC_d = \frac{\sum_{c=1}^{m} p_c \cdot SOC_{cd}}{\sum_{c=1}^{m} p_c} \qquad (3)$$

where p_c is the component percentage and *m* is the number of components present in the MU under consideration. All soil components (including minor as well as major) were taken into account when calculating the amount of organic matter and bulk density within each map unit, for each horizon (Bridgham *et al*., 2006). This method is time-consuming and detailed, but allowed the representation of a soil to be much more accurate than considering only a single or dominant component. Within a separate output file, I listed all 8714 map units along with the percentage of each component within them, and any horizon data that were missing. For most map units within the database, the components added up to 100%. For those that did not, these components were considered partially missing. CoBluCarb had as output the organic carbon stock, SOC_d, at depth increments of 5 cm, as well as the calculated carbon density (OCD_d).

II.2.4 Evaluation of CoBluCarb

For evaluation and comparison, I regressed the carbon density values from our database against those found at the same spatial location as sourced from Ouyang and Lee 2014 and Chmura *et al.,* 2003. These two literature reviews contain the most expansive published compilation of carbon density for wetlands, to our knowledge. Each is based on a number of field samples, compiled from a number of other research articles. Upon initial investigation, I found the correlation between our dataset and these values to be potentially significant though relatively low (Fig. 1).

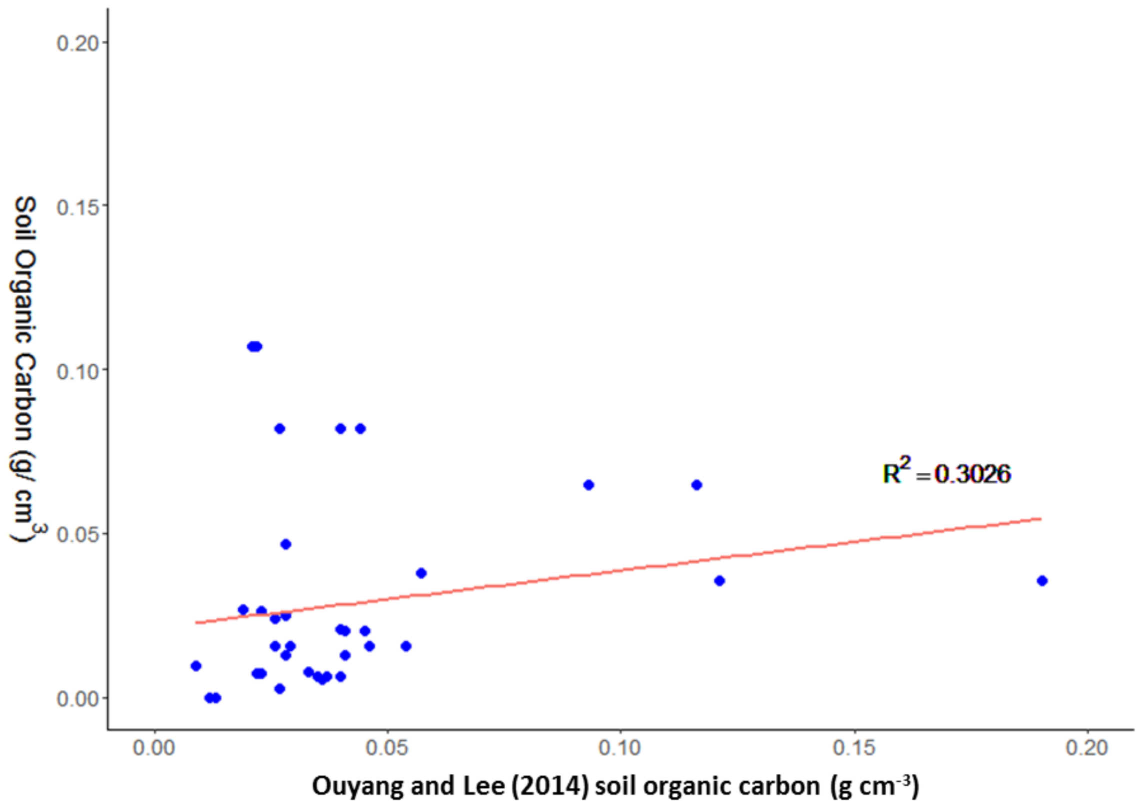

Figure 1. *Linear regression of literature-derived values versus CoBluCarb values*

The scatter and variance were likely attributable to at least four sources of error: (1) the locations of the literature-derived field samples were coarse and imprecise; (2) the literature-derived values were not accurate to the depth range at which they were acquired, and yet carbon density varied considerably with depth as shown in our database; (3) the literature-derived values were acquired using many different methods, with variable degrees of accuracy and precision; (4) soils in general, and tidal wetland soils in particular, have a high degree of spatial heterogeneity, with variable soil typology, density, and SOC content within only a few meters of distance. These literature values are also primarily representative of soil surface samples (Morris *et al.,* 2012), do not have confidence intervals for the quality of the data originating from their source datasets, and do not contain sums of the soil organic carbon totals across the sampled depths. Based on these findings (Fig. 1), I concluded that CoBluCarb presented herein was likely the more spatially accurate, depth-explicit, methodologically consistent, and widely applicable stock estimate for the continental US wetlands.

II.2.5 *Error in CoBluCarb*

CoBluCarb does have quantifiable error in the form of both bias and variance. These errors could source from SSURGO, NWI, their spatial combination, or calculations from Eq. 1-3 performed to arrive at the output values. There are currently other groups that have been conducting work on bias and variance in the carbon density values, and two separate lines of on-going and extensive inquiry exist. The details of these studies are numerous and are thus not included here due to space constraints, however I provide the following review of their findings to date.

Some of these studies have indicated that SSURGO contains upward bias in its reported bulk density values, or BD from Eq. 1. This line of work has attempted to correct this bias by regressing BD from SSURGO against a large compilation of BD data from wetland cores, similar to those found in Morris et al. (2016), and then adjusting the BD values downward to better fit the core data. This line of work has also created a much larger validation dataset (of carbon density) than that contained in Ouyang and Lee (2014) and Chmura (2003) alone. The early results from this approach show that after correcting this bias, the relationship between SSURGO itself and the larger validation dataset is not substantially improved. The headline result of this effort, at least initially, is likely to be that the bias-corrected mean carbon density value from SSURGO yields just as good of a fit to the validation data as for that from any one specific location. Another study (Hinson et al., 2017) has found that that the variance among carbon density values in CoBluCarb, which are sourced from SSURGO, can be constrained within specific categorical and geographic groupings. This line of work has attempted to regress the SSURGO-derived carbon density from CoBluCarb against an array of spatially-identifiable factors, including latitude, longitude, precipitation, temperature, salinity, tidal range, etc. This line of work has also tested whether specific categories of SSURGO-derived carbon density data contain lower variance than the overall mean, for example by categorizing by wetland types, salinity types, coasts, etc. The source datasets used to categorize and aggregate the data include NWI, CAF, and PRISM datasets (Oregon State PRISM Climate Group, 2017). The early results from this approach show that the variability among the carbon density values are related to, and constrained by, several spatial and categorical factors. The headline result of this effort, at least initially, is likely to be carbon density can be known with a specific amount of inherent error, given user information on location or category.

The difference between these two on-going investigations into error highlights the difference between bias and variance. The first effort addresses bias relative to an expanded validation dataset (accuracy) and the second addresses constraining the variance of the dataset across factors that are known (precision).

II.2.6 *Further Investigation of Bias*

To understand whether there is a mean bias to a dataset requires knowledge of the 'truth'. It is a difficult proposition to know whether SSURGO-derived CoBluCarb or compiled literature values are more accurate, other than to compare them to additional measures of the 'truth'. As described above in the section entitled *Evaluation Dataset,* the literature-derived field samples appeared to suffer from greater inconsistency than the SSURGO-derived CoBluCarb dataset, in four specific ways, but whether this translates to greater bias requires comparisons to additional datasets.

First, since the literature-derived values were not depth-explicit, I sought to further compare CoBluCarb with two recent studies that did provide depth-explicit data. For multiple sample depths from these studies, I matched CoBluCarb carbon densities at each depth and then regressed them. In Yando et al. (2016), Louisiana, Central Texas, and Florida locations were included, though I eliminated the Florida sites as they were in Monroe County (see main text for specific problems in that location). In Doughty et al. (2016), Florida locations were included. The depth-dependent correlation between CoBluCarb and the values in each study were strongly or moderately correlated (r^2 = 0.7818 with a p value = 0.019 and r^2 = 0.6191 with a p value = 0.2131, respectively), suggesting that the lack of depth-explicit data was likely a problem for the Ouyang and Lee (2014) or Chmura (2003) literature-derived values. However, when the depth-explicit data from Yando et al. (2016) and Doughty et al. (2016) were jointly regressed against CoBluCarb, the correlation was poor (r^2 = 0.0049 with a p value = 0.8462). In other words, while the variation of values from a single study may have a tight fit with respect to CoBluCarb, the slope of that fit may be different than the slope of fit between a second study and CoBluCarb. This example highlights how error can be propagated when using compiled literature values to create validation datasets – the shunting of multiple original datasets into a single amalgamation or compilation ignores numerous inconsistencies and the mean biases among them. As mentioned previously, one advantage to SSURGO-derived CoBluCarb values is that the NRCS-mandated methodology and depth-explicit data was consistently recorded across the many field sample locations.

A second source of bias could lie within CoBluCarb itself, created through the cascading of errors among the multiple datasets that were intersected together during its creation. In particular, the SSURGO data available from the NRCS is sourced from an original NRCS soil core dataset, here called the 'pedon' data. The pedon data itself is described in the first paragraph of the *USDA SSURGO Dataset* section of the main text, but as the second paragraph describes this information was converted by the NRCS into MUs and components by specialist interpretation – leaving open the possibility of introducing bias and variation into SSURGO. There is no known publication or report determining how error may be introduced between the pedon data and its transference to the SSURGO format. While there are other options that could have been used in the creation of CoBluCarb such as the State Soil Geographic Database (STATSGO) or the National Soil Geographic database (Bliss *et al,.*1995; West *et al.,* 2008; Guo *et al.,* 2006), these are coarser datasets. STATSGO and SSURGO have been compared in terms of soil carbon (Zhong and Xu 2011; Bliss *et al.,* 1995; Davidson and Lefebvre

1993) but the finer scale data is important when considering soil organic carbon variability. To investigate these types of cascading errors, I examined the USDA pedon data against the CoBluCarb output, in terms of BD and OM. Of the 18 total pedons in tidal wetlands, none were able to be related to the SSURGO data (they lacked BD and OM data) and were unusable to analyze for accuracy in SSURGO. This is another avenue for future research and potential to further the accuracy of large scale soil databases.

Like the literature-derived carbon density values, CoBluCarb is another compilation of field data that is based on coring procedures though it is also the most extensive compilation of such data that is spatially-explicit. On average, it presents lower carbon density values than those found in Ouyang and Lee (2014) or Chmura (2003) (Fig. 1). Location-specific and depth-explicit information is crucial for SOC determination due to the high variability in coastal wetland soils. Consistent sampling methodology is crucial to minimize internal database bias, as well as internal variability. CoBluCarb meets these criteria. It is more extensive and its errors are better-quantified, as compared to previous efforts. The database is not intended to provide precise answers within a few meters, but rather to provide a comprehensive overview of estuarine basins and specific wetland types, to show potential trends and patterns that can aid in the management and conservation of these areas. For these intended purposes, CoBluCarb is a reasonable and quantified analysis tool for managers and scientists considering coastal wetland SOC.

II.2.7 *Handling Missing SSURGO Data*

There are 11 different cases of missing values found in the original SSURGO database based on components and horizons (Table 1).

For full components missing from the SSURGO database, the percentage of the missing component was considered "Null," such that the other components with values were rescaled to cover the missing percentage and make the total of the components to equal 100% for the map unit (Cases B, M, U, and Y).

In another type of missing data in the SSURGO database, there were components that were not considered traditional soils. These include map units called "urban areas," "dumps," or "water,", yet the NWI dataset categorized as wetlands. For the purpose of this study, I took the NWI data as the priority definition of what should be considered a wetland. For these soils, the assumption was made that the soil organic carbon was negligible. These components are all equal to "Null" since there is no information, horizon or otherwise, within these components (Cases V and Z).

Within the components on the horizon level, there were cases in which a single or multiple horizons were missing (horizon is missing in entirety), or a single value was missing (organic matter or bulk density) within the horizon. While these issues could be due to a multitude of reasons such as inability to sample, compromised sample, or only a very thin horizon, the values could potentially have an influence on the resulting carbon stocks and densities. If only one horizon or any portion of that horizon was missing within a component, the horizon was still considered within the component, but the values were considered zero. If the bulk density or the organic matter fraction were missing, the value was also assigned zero, which resulted in a zero in the overall carbon calculation for that horizon. Since in these instances the placeholders for each potential value were listed, even though there were no data values entered, they were still considered within the calculations. Typically, these horizons were very thin or at the bottom of a component's depth profile, well below the range of a typical user inquiry, or below the depth where the carbon profile leveled out to a background quantity (Cases M and L). There was only one soil in the entire database that had horizon information without depth indications (Case F). Since, depth is an integral portion of this project, but the information was not there, the soil carbon value was determined as zero.

In the case when there was a "Null" value in a middle horizon that should be considered in the calculations, the "sandwich method" was enforced. In this method, the two horizons surrounding the missing horizon were averaged to give the missing horizon a value.

As for Case A, since the values are expressed in the totals and the information is complete, the case was viewed as if the values reflected were true. Therefore, if the values created carbon as 0, then the value for the soil organic carbon for that horizon or depth was 0. For Case X, the values were considered Null because even though the horizon percentages were complete, there were no values to correspond with the percentages.

Table 1. *A detailed description of the types of missing values considered within the SSURGO dataset*

Missing Data Type	Description	SOC Value
A	All horizons have properties & soil components total percentages are complete	Integer or 0
B	Soil components total percentage is not 100% complete	NULL
L	Incomplete horizon in one or more components & soil components total percentage is 100%	0
M	Incomplete horizon in one or more components & soil components percentage is between 0-100%	0, THEN NULL
T	No horizon properties listed in any component but components total percentage is (100%)	NULL
U	No horizon properties listed in any component & components total percentage is not 100%	NULL
V	No horizon properties listed in any component and soil components total percentage is 0%	NULL
X	No horizon properties, but soil components total percentage is 100% complete	NULL
Y	No horizon properties and soil components total percentage is not 100% complete	NULL
Z	No horizon properties or soil components total percentages	NULL
F	Horizon properties but no depth	0

II.2.8 *Spatial Distribution of Tidal Wetland Carbon*

For visual presentation, I populated the NWI records/polygons with the calculated values from CoBluCarb. The final spatial database contained the following data at 5-cm increments in the soil profile, for each polygon: total SOC amount (total mass within an area), and carbon density (g cm^{-3}).

The sum of the total SOC amounts and the area-weighted average of the carbon densities of the polygons were then calculated according to their wetland type, as well as the state and the estuary in which they existed. To define estuarine extents, I used spatial boundaries from the National Oceanic and Atmospheric Administration's Coastal Assessment Framework (CAF), containing a total of 115 Estuarine Drainage Areas (EDAs) and 199 Coastal Drainage Areas (CDAs). Here, I present only the results from the EDAs and summarize all of the CDAs with a single presented value. From here forward in the current presentation, the results are reported across ranges at a 0-15 cm depth increment for SOC management and restoration purposes, at a 0-100 cm depth increment for SOC conservation purposes, and at the 100 cm depth itself for general scientific purposes.

For the two depth intervals of 0-15 and 0-100 cm, I present both low and high boundary limits. The low-limit was based only upon the SOC quantities within the database, excluding any cases of missing SSURGO data. This value should be viewed as a conservative estimate. The high-limit was based on an assumption that these missing cases were similar to the known cases within the same geographical extent on an area-weighted basis. This value should be seen as a liberal estimate. For the area-weighted average carbon density across each of the two depths, as well as at the 100 cm depth itself, I summarized only the individual polygons/ records with complete information.

II.3 Results

Across the tidal wetland soils area of the continental United States, the data is heterogeneous though the mode of all soils' organic carbon densities is ~0.05 g cm^{-3}. The average density across all tidal wetlands was 0.071 g cm^{-3} across 0-15, 0.055 g cm^{-3} across 0-100, and 0.040 g cm^{-3} at the 100 cm depth. There are additional peaks in histograms at higher densities, particularly for the upper 0-15 cm of depth (Fig. 2).

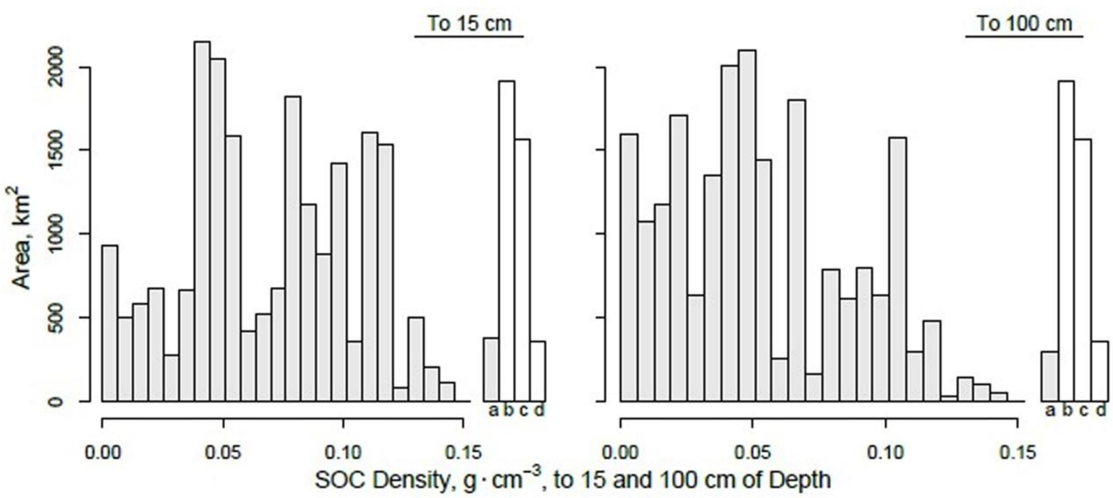

Figure 2. *Histogram showing the total wetland area for each carbon density across all MU soil types and locations in the upper 15 cm and the upper 100 cm. Bars a-d denote the amount of missing area for special cases: (a) is the excess area specific SOC densities greater than 0.15 (b) is the total area missing for Monroe county in Florida, located in the Everglades, (c) is the total area for all wetlands with missing data in any other location besides Monroe County in South Florida, and (d) the 355 km^2 of wetland polygons that do not overlap any SSURGO data. Columns that show carbon density are shaded and columns with area that has missing carbon information are unshaded (extra columns b-d).*

Unique trends in the SOC density of soils based on their area of coverage can be seen across different categorizations of wetland type and coasts. The carbon density in emergent vegetation wetlands is somewhat normally distributed by areal coverage and notably covers much larger areas than the other vegetation types (Fig. 3).

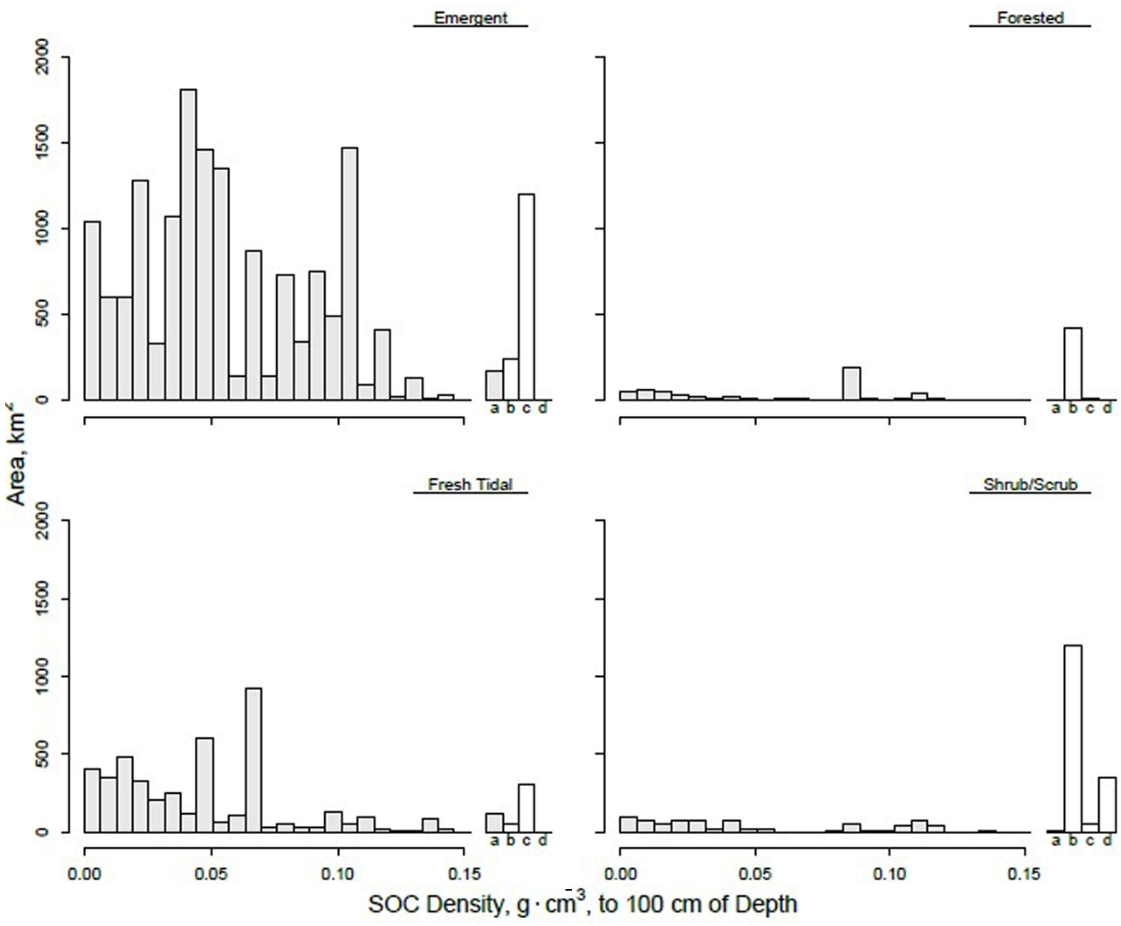

Figure 3. *Histogram showing the total wetland area for each carbon density across the specific wetland categorizations at a range of 0-100 cm depth interval. In the upper 15 cm, the results are similar and show the same trends as 0-100 cm. Refer to Figure 2 caption for columns a-d.*

The distribution across the Gulf Coast is also somewhat normal, while the East Coast is more bi-modal (Fig. 4). The West Coast has relatively little area of tidal wetland soils. Most SOC is stored in estuarine emergent wetlands (Table 2), which contain over three times the SOC of the next closest wetland type (freshwater tidal wetlands), and exceeded all other types in storage due to their greater geographic extent.

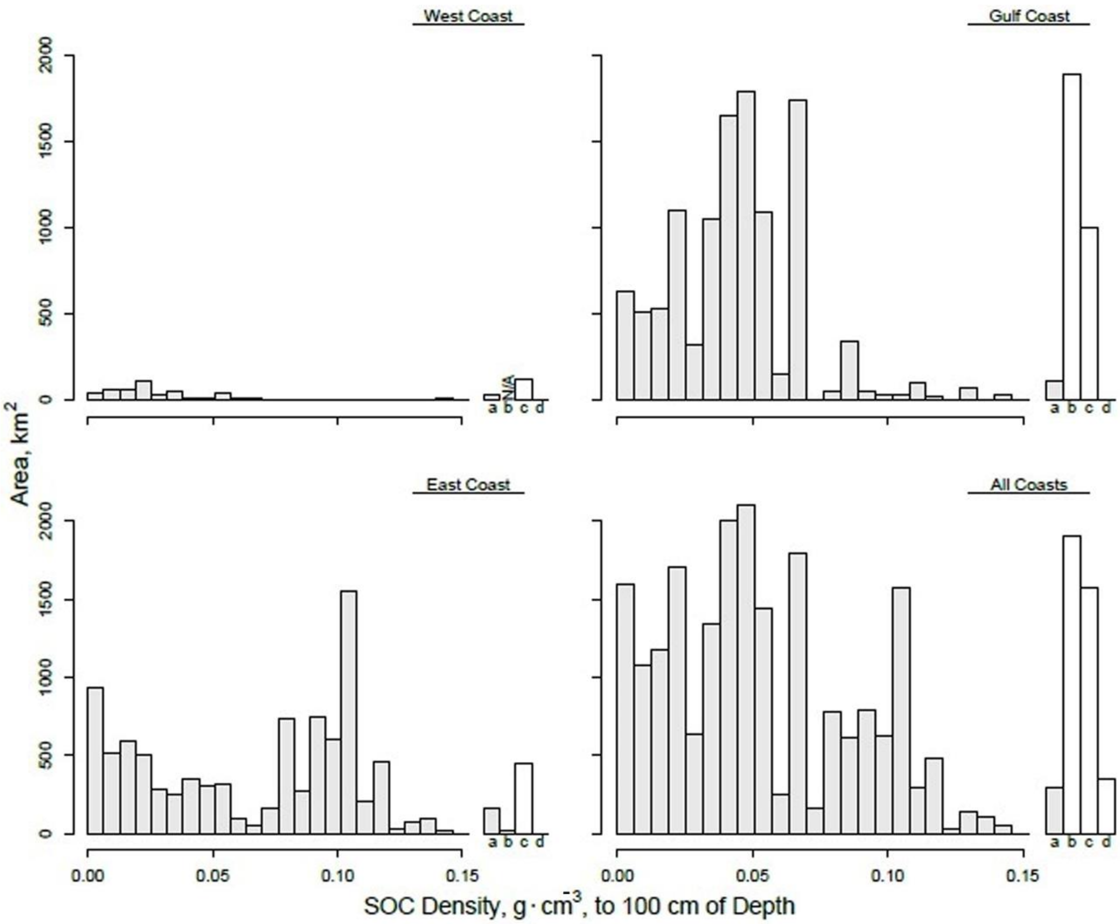

Figure 4. *Histogram showing the total wetland area for each carbon density across each coast from 0-100 cm depth increments. In the upper 15 cm, the results are similar and show the same trends as 0-100 cm. Refer to Fig. 2 caption for columns a-d.*

Soil in forested and emergent vegetation wetlands had a higher area-weighted average carbon density (0.075 and 0.074 g cm^{-3} respectively) in the upper layers of soils (top 0-15 cm), than the other wetland types. However, this disparity in densities was less pronounced when comparing SOC stored in 0-100 cm depth increment.

Table 2. *Soil organic carbon by wetland type and coast. Lo identifies quantities that were based on polygons for which SSURGO data was not completely missing; Hi extrapolates the Lo quantities to all associated polygons assuming that the densities for all polygons are adequately represented by the (Lo) density found from polygons having SSURGO information.*

	Area (km^2)		Stock: 0-15 cm (Tg)		Stock: 0-100 cm (Tg)		Density (g/cm^3)		Density (g/cm^3)
Category	Lo	Hi	Lo	Hi*	Lo	Hi*	0-15 cm	0-100 cm	At 100 cm
Estuarine Emergent	15272.5	16985.4	169.0	188.0	859.9	956.3	0.074	0.056	0.043
Estuarine Forested	474.2	925.7	5.3	10.4	27.2	53.2	0.075	0.057	0.044
Estuarine Shrub-Scrub	788.5	2073.2	7.9	20.7	40.1	105.5	0.067	0.051	0.033
Freshwater Tidal	4577.8	4961.7	43.1	46.7	225.4	244.3	0.063	0.049	0.042
East	9329.2	9818.5	113.6	119.6	623.6	656.3	0.081	0.067	0.058
Gulf	11344.7	14559.9	108.0	138.6	511.4	656.4	0.063	0.045	0.030
West	439.0	567.5	3.7	4.7	17.6	22.8	0.056	0.040	0.033

* Refer to table consistency section in Discussion section

The average organic carbon density decreased with greater soil depth for all wetland types (Fig. 5a). However, only the estuarine emergent type displayed a noticeably decreasing standard deviation with greater depth (Fig. 5b), with its inflection point at around 25-30 cm of depth (roughly expected as the rooting depth for herbaceous cover).

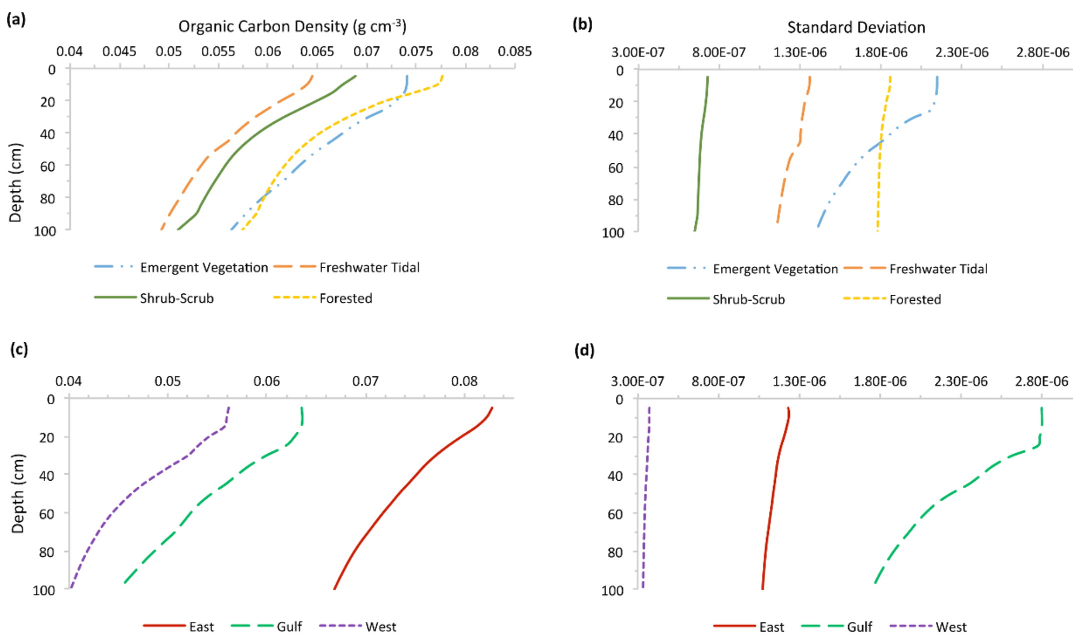

Figure 5. *Vertical profiles in the upper 100 cm of soil of (a) average area weighted carbon densities for each wetland type, (b) standard deviation of the average area weighted carbon density for each wetland type, (c) average area weighted carbon density for each coast, and (d) standard deviation of the average area weighted carbon density for each coast.*

When total SOC amount is viewed by coastal region (Table 2), the East Coast (113.6 - 119.6 Tg) and the Gulf Coast (108.0 - 138.6 Tg) were found to be similar at the range 0-15 cm of depth, though the area of wetlands on the Gulf Coast was much larger (approximately 30% greater area considering the high estimate). Even in the 0-100 cm depth interval, the range maxima were similar, though the range was much larger for the Gulf Coast (511.4 - 656.4 Tg) than the East Coast (623.6 - 656.3 Tg). The discrepancies between the high and low values for a given range are direct reflections of the lack of data for wetland areas, for example and in particular Monroe County on the Gulf Coast of Florida within the Everglades. The Monroe County area is important for calculating the total carbon stock for the Gulf Coast, but since there was no soil data in SSURGO, the difference in the high and low value was reflected accordingly. If accurate soil information was obtained for this specific area, the Gulf Coast would more precisely reveal the current carbon stock. The average organic carbon densities for all three coasts decreased with greater depth in similar fashion (Fig. 5c). However, the Gulf Coast organic carbon density was the most variant, particularly at shallow depths (Fig. 5d). When viewed by state (Table 3), Louisiana is seen to have the most SOC across the 15 cm and 100 cm depth intervals, due to both relatively high density values and its large expanse of wetlands. Florida had a large range in total SOC from 0-100 cm of depth (130.6 - 237.3 Tg), due to large areas of wetlands missing vital measurements for calculating soil organic carbon, notably in Monroe County and the Everglades. At 0-15 cm of depth, Georgia had the greatest area weighted average density (0.106 g cm^{-3}) while across 0-100 cm of depth, Mississippi had the greatest (0.096 g cm^{-3}). Many of the states had relatively similar area weighted average density values, falling within 0.06 - 0.09 g cm^{-3} for upper 0-15 cm of depth and 0.04 - 0.06 g cm^{-3} for 0-100 cm depth increment. Texas had the second lowest (0.036 g cm^{-3} for 0-15 cm, and 0.018 g cm^{-3} for 0-100 cm) while also having a large area of tidal wetlands.

When total SOC amount is viewed by estuary (Table 4, Fig. 6), there is much variability. The Atchafalaya/Vermilion Bays complex had the greatest total SOC in the upper 100 cm of depth (116.0 - 125.3 Tg) and Chesapeake Bay was second (89.8 - 95.9 Tg). Neither of these estuaries had the highest area weighted density though. The Merrimack River estuary had the greatest density in the upper 0-15 cm and 0-100 cm (respectively, 0.109 and 0.098 g cm^{-3}). Florida Bay had the second greatest density (respectively, 0.087 and 0.068 g cm^{-3}). Many Southern California and South Texas estuaries had quite low total SOC amounts (Mission Bay, California was the lowest of all estuaries) and many of their densities were quite low as well (San Pedro, California was the lowest at 0.002 g cm^{-3}). Several open-ocean CDAs had still lower values, primarily on the US West Coast.

Table 3. *Soil organic carbon by state. Lo identifies quantities that were based on polygons for which SSURGO data was not completely missing; Hi extrapolates the Lo quantities to all associated polygons assuming that the densities for all polygons are adequately represented by the (Lo) density found from polygons having SSURGO information.*

	Area (km^2)			Stock to 0-15 cm (Tg)			Stock to 0-100 cm (Tg)			Density (g/cm^3)			Density (g/cm^3)
	Lo	-	Hi	Lo	-	Hi*	Lo	-	Hi*	0-15 cm	-	0-100 cm	At 100 cm
AL	112.9	-	121.1	0.6	-	0.7	5.1	-	5.4	0.037	-	0.045	0.045
CA	178	-	266	2.0	-	2.9	10.6	-	15.8	0.073	-	0.059	0.051
CT	56.2	-	60.5	0.8	-	0.9	4.0	-	4.3	0.096	-	0.072	0.051
DC	0.3	-	0.5	0.0	-	0.0	0.0	-	0.0	0.012	-	0.006	0.003
DE	326	-	335.1	3.1	-	3.2	17.7	-	18.2	0.063	-	0.054	0.066
FL	2516	-	4570.4	27.4	-	49.8	130.6	-	237.3	0.073	-	0.052	0.036
GA	1784.8	-	1825.2	28.3	-	28.9	148.3	-	151.7	0.106	-	0.083	0.076
LA	7914.5	-	9057.4	77.2	-	88.4	365.4	-	418.1	0.065	-	0.046	0.030
MA	206.6	-	215.4	2.8	-	2.9	16.0	-	16.7	0.089	-	0.078	0.070
MD	1027.3	-	1093.7	11.9	-	12.6	61.0	-	64.9	0.077	-	0.059	0.049

Table 3 Continued

	Area (km^2)			Stock to 0-15 cm (Tg)			Stock to 0-100 cm (Tg)			Density (g/cm^3)			Density (g/cm^3)
	Lo	-	Hi	Lo	-	Hi*	Lo	-	Hi*	0-15 cm	-	0-100 cm	At 100 cm
ME	124.5	-	140.4	1.2	-	1.3	6.6	-	7.4	0.062	-	0.053	0.052
MS	283.2	-	290.7	4.1	-	4.2	27.2	-	27.9	0.097	-	0.096	0.082
NC	1111.8	-	1145.5	15.6	-	16.1	91.3	-	94.1	0.093	-	0.082	0.057
NH	24.9	-	26.8	0.3	-	0.4	1.9	-	2.1	0.09	-	0.078	0.070
NJ	924	-	954.8	12.3	-	12.7	77.3	-	79.9	0.089	-	0.084	0.085
NY	74.9	-	134	1.0	-	1.8	5.7	-	10.2	0.089	-	0.076	0.066
OR	114.1	-	124.7	0.7	-	0.7	3.6	-	4.0	0.04	-	0.032	0.027
PA	1.6	-	2.4	0.0	-	0.0	0.1	-	0.1	0.045	-	0.037	0.065
RI	15.3	-	15.8	0.2	-	0.2	0.7	-	0.8	0.081	-	0.048	0.035
SC	1940.5	-	2021.6	18.9	-	19.7	108.1	-	112.6	0.065	-	0.056	0.049

Table 3 Continued

	Area (km^2)			Stock to 0-15 cm (Tg)			Stock to 0-100 cm (Tg)			Density (g/cm^3)			Density (g/cm^3)
	Lo	-	Hi	Lo	-	Hi*	Lo	-	Hi*	0-15 cm	-	0-100 cm	At 100 cm
TX	1190.1	-	1252.5	6.4	-	6.7	21.1	-	22.2	0.036	-	0.018	0.009
VA	1038.7	-	1114.6	9.6	-	10.3	46.8	-	50.2	0.062	-	0.045	0.037
WA	146.9	-	176.8	1.0	-	1.2	3.4	-	4.1	0.047	-	0.023	0.015

*Refer to table consistency section in text

Table 4. Soil organic carbon by estuarine drainage area (EDA). All coastal drainage areas (CDAs) are consolidated as one. Lo identifies quantities that were based on polygons for which SSURGO data was not completely missing; Hi extrapolates the Lo quantities to all associated polygons assuming that the densities for all polygons are adequately represented by the (Lo) density found from polygons having SSURGO information.

	Area (km^2)			Stock to 0-15 cm (Tg)			Stock to 0-100 cm (Tg)			Density (g/cm^3)			Density (g/cm^3)
	Lo	-	Hi	Lo	-	Hi*	Lo	-	Hi*	0-15 cm	-	0-100 cm	At 100 cm
Albemarle Sound	349.6	-	357.0	5.2	-	5.3	33.1	-	33.8	0.099	-	0.095	0.073
Alsea River	2.6	-	3.3	0.0	-	0.0	0.1	-	0.1	0.052	-	0.033	0.021
Altamaha River	177.4	-	180.6	2.0	-	2.0	7.6	-	7.8	0.074	-	0.043	0.037
Apalachee Bay	200.5	-	208.0	2.2	-	2.3	9.3	-	9.7	0.072	-	0.047	0.029
Apalachicola Bay	130.1	-	135.0	1.2	-	1.3	6.8	-	7.1	0.063	-	0.053	0.048
Aransas Bay	93.2	-	102.9	0.1	-	0.2	0.8	-	0.9	0.010	-	0.009	0.008
Atchafalaya/Vermilion Bays	2282.4	-	2465.5	20.8	-	22.4	116.0	-	125.3	0.061	-	0.051	0.042
Barataria Bay	750.7	-	1151.0	6.7	-	10.3	37.8	-	58.0	0.059	-	0.050	0.033
Barnegat Bay	53.7	-	55.5	0.6	-	0.6	3.3	-	3.4	0.070	-	0.061	0.060

Table 4 Continued

	Area (km^2)			Stock to 0-15 cm (Tg)			Stock to 0-100 cm (Tg)			Density (g/cm^3)			Density (g/cm^3)
	Lo	-	Hi	Lo	-	Hi*	Lo	-	Hi*	0-15 cm	-	0-100 cm	At 100 cm
Biscayne Bay	99.9	-	131.8	1.2	-	1.6	7.0	-	9.2	0.082	-	0.070	0.054
Blue Hill Bay	4.4	-	4.5	0.0	-	0.0	0.2	-	0.2	0.065	-	0.047	0.027
Bogue Sound	89.0	-	95.1	1.1	-	1.2	5.0	-	5.3	0.085	-	0.056	0.034
Brazos River	113.9	-	116.6	0.8	-	0.8	2.5	-	2.6	0.044	-	0.022	0.008
Breton/Chandeleur Sound	1120.3	-	1247.1	10.0	-	11.2	48.9	-	54.4	0.060	-	0.044	0.025
Broad River	293.3	-	308.8	4.7	-	5.0	27.3	-	28.7	0.107	-	0.093	0.080
Buzzards Bay	23.1	-	23.8	0.3	-	0.3	1.7	-	1.8	0.087	-	0.075	0.069
Calcasieu Lake	625.9	-	667.4	8.7	-	9.3	27.2	-	29.0	0.093	-	0.043	0.011
Cape Cod Bay	49.7	-	52.5	0.6	-	0.6	3.5	-	3.7	0.081	-	0.070	0.063
Cape Fear River	42.9	-	44.5	0.6	-	0.6	3.6	-	3.8	0.089	-	0.085	0.053

Table 4 Continued

	Area (km^2)			Stock to 0-15 cm (Tg)			Stock to 0-100 cm (Tg)			Density (g/cm^3)			Density (g/cm^3)
	Lo	-	Hi	Lo	-	Hi*	Lo	-	Hi*	0-15 cm	-	0-100 cm	At 100 cm
Casco Bay	7.7	-	9.5	0.0	-	0.1	0.3	-	0.4	0.037	-	0.044	0.063
Charleston Harbor	139.5	-	150.7	1.0	-	1.1	4.8	-	5.2	0.049	-	0.035	0.030
Charlotte Harbor	217.8	-	222.5	2.4	-	2.4	10.9	-	11.1	0.073	-	0.050	0.017
Chesapeake Bay	1546.4	-	1651.0	17.3	-	18.4	89.8	-	95.9	0.074	-	0.058	0.048
Chincoteague Bay	109.6	-	118.6	1.0	-	1.1	4.4	-	4.8	0.063	-	0.041	0.031
Choctawhatchee Bay	37.3	-	38.7	0.3	-	0.3	1.9	-	2.0	0.057	-	0.052	0.048
Columbia River	94.0	-	104.9	0.5	-	0.5	2.1	-	2.4	0.034	-	0.023	0.018
Coos Bay	5.9	-	7.5	0.0	-	0.1	0.3	-	0.4	0.055	-	0.051	0.046
Coquille River	1.2	-	1.4	0.0	-	0.0	0.1	-	0.1	0.048	-	0.044	0.040
Corpus Christi Bay	32.0	-	35.5	0.0	-	0.0	0.2	-	0.2	0.008	-	0.006	0.006
Damariscotta River	0.9	-	1.0	0.0	-	0.0	0.0	-	0.1	0.070	-	0.052	0.044

Table 4 Continued

	Area (km^2)			Stock to 0-15 cm (Tg)			Stock to 0-100 cm (Tg)			Density (g/cm^3)			Density (g/cm^3)
	Lo	-	Hi	Lo	-	Hi*	Lo	-	Hi*	0-15 cm	-	0-100 cm	At 100 cm
Delaware Bay	673.2	-	699.0	7.7	-	8.0	46.9	-	48.7	0.076	-	0.070	0.075
Delaware Inland Bays	25.3	-	26.2	0.3	-	0.3	1.1	-	1.1	0.067	-	0.043	0.040
Drakes Estero	0.3	-	2.1	0.0	-	0.0	0.0	-	0.1	0.034	-	0.026	0.020
East Mississippi Sound	165.2	-	170.2	1.7	-	1.8	11.0	-	11.4	0.069	-	0.067	0.055
Eel River	1.6	-	1.6	0.0	-	0.0	0.0	-	0.0	0.039	-	0.014	0.005
Englishman/Machias Bay	6.1	-	7.3	0.1	-	0.1	0.3	-	0.4	0.073	-	0.056	0.047
Florida Bay	23.9	-	330.3	0.3	-	4.3	1.6	-	22.5	0.087	-	0.068	0.047
Galveston Bay	335.4	-	347.0	2.1	-	2.1	6.5	-	6.7	0.041	-	0.019	0.007
Gardiners Bay	2.8	-	16.5	0.0	-	0.1	0.1	-	0.4	0.051	-	0.026	0.006

Table 4 Continued

	Area (km^2)			Stock to 0-15 cm (Tg)			Stock to 0-100 cm (Tg)			Density (g/cm^3)			Density (g/cm^3)
	Lo	-	Hi	Lo	-	Hi*	Lo	-	Hi*	0-15 cm	-	0-100 cm	At 100 cm
Grays Harbor	39.7	-	43.4	0.3	-	0.3	1.0	-	1.1	0.053	-	0.026	0.017
Great Bay	9.0	-	10.2	0.1	-	0.1	0.5	-	0.5	0.077	-	0.052	0.037
Great South Bay	35.6	-	69.8	0.4	-	0.9	2.8	-	5.4	0.084	-	0.078	0.074
Hampton Harbor	14.3	-	15.0	0.2	-	0.2	1.3	-	1.3	0.094	-	0.089	0.086
Hudson River/Raritan Bay	86.2	-	90.2	1.1	-	1.1	6.2	-	6.5	0.085	-	0.072	0.064
Humboldt Bay	1.4	-	4.7	0.0	-	0.0	0.0	-	0.1	0.041	-	0.014	0.006
Indian River	137.5	-	143.6	0.9	-	0.9	4.1	-	4.2	0.044	-	0.029	0.020
Kennebec/Androscoggin River	26.6	-	31.8	0.3	-	0.3	1.5	-	1.8	0.070	-	0.056	0.049
Klamath River	0.0	-	0.4	0.0	-	0.0	0.0	-	0.0	0.031	-	0.013	0.007

Table 4 Continued

	Area (km^2)			Stock to 0-15 cm (Tg)			Stock to 0-100 cm (Tg)			Density (g/cm^3)			Density (g/cm^3)
	Lo	-	Hi	Lo	-	Hi*	Lo	-	Hi*	0-15 cm	-	0-100 cm	At 100 cm
Long Island Sound	60.1	-	72.6	0.9	-	1.0	4.3	-	5.2	0.095	-	0.072	0.052
Lower Laguna Madre	49.7	-	60.0	0.0	-	0.0	0.2	-	0.2	0.005	-	0.004	0.003
Maryland Inland Bays	19.4	-	21.2	0.2	-	0.3	1.0	-	1.1	0.085	-	0.051	0.051
Massachusetts Bay	25.7	-	27.7	0.3	-	0.4	2.0	-	2.2	0.089	-	0.078	0.072
Matagorda Bay	245.2	-	255.0	1.3	-	1.3	4.3	-	4.4	0.034	-	0.017	0.007
Mermentau River	735.1	-	776.6	9.1	-	9.6	30.5	-	32.2	0.082	-	0.041	0.014
Merrimack River	10.9	-	11.0	0.2	-	0.2	1.1	-	1.1	0.109	-	0.098	0.089
Mission Bay	0.0	-	0.2	0.0	-	0.0	0.0	-	0.0	0.007	-	0.002	0.000
Mississippi River	276.2	-	423.1	2.2	-	3.4	10.3	-	15.8	0.053	-	0.037	0.026
Mobile Bay	65.5	-	72.6	0.5	-	0.5	2.6	-	2.9	0.046	-	0.04	0.038
Monterey Bay	6.0	-	6.7	0.0	-	0.0	0.1	-	0.1	0.012	-	0.011	0.011

Table 4 Continued

	Area (km^2)			Stock to 0-15 cm (Tg)			Stock to 0-100 cm (Tg)			Density (g/cm^3)			Density (g/cm^3)
	Lo	-	Hi	Lo	-	Hi*	Lo	-	Hi*	0-15 cm	-	0-100 cm	At 100 cm
Morro Bay	0.2	-	2.0	0.0	-	0.0	0.0	-	0.0	0.017	-	0.016	0.002
Muscongus Bay	2.5	-	3.1	0.0	-	0.0	0.1	-	0.2	0.076	-	0.055	0.047
Narragansett Bay	14.7	-	15.0	0.2	-	0.2	0.8	-	0.8	0.084	-	0.056	0.043
Narraguagus Bay	9.6	-	10.4	0.1	-	0.1	0.6	-	0.6	0.077	-	0.062	0.055
Nehalem River	5.8	-	6.1	0.0	-	0.0	0.2	-	0.3	0.05	-	0.043	0.040
Netarts Bay	0.8	-	0.9	0.0	-	0.0	0.0	-	0.0	0.055	-	0.051	0.049
New Jersey Inland Bays	421.4	-	431.0	6.0	-	6.2	39.1	-	39.9	0.096	-	0.093	0.095
Newport Bay	0.2	-	1.4	0.0	-	0.0	0.0	-	0.0	0.004	-	0.002	0.042
New River	25.7	-	27.7	0.3	-	0.3	1.8	-	1.9	0.075	-	0.07	0.000

Table 4 Continued

	Area (km^2)			Stock to 0-15 cm (Tg)			Stock to 0-100 cm (Tg)			Density (g/cm^3)			Density (g/cm^3)
	Lo	-	Hi	Lo	-	Hi*	Lo	-	Hi*	0-15 cm	-	0-100 cm	At 100 cm
North/South Santee Rivers	160.9	-	163.5	1.3	-	1.3	7.4	-	7.5	0.052	-	0.046	0.074
North Ten Thousand Islands	237.6	-	924.9	3.0	-	11.6	17.9	-	69.6	0.084	-	0.075	0.044
Ossabaw Sound	219.6	-	222.4	3.1	-	3.2	17.7	-	18.0	0.095	-	0.081	0.073
Pamlico Sound	634.2	-	648.5	8.7	-	8.9	50.0	-	51.1	0.091	-	0.079	0.058
Passamaquoddy Bay	2.0	-	3.3	0.0	-	0.0	0.1	-	0.1	0.06	-	0.042	0.031
Penobscot Bay	5.8	-	6.2	0.1	-	0.1	0.3	-	0.4	0.073	-	0.059	0.052
Pensacola Bay	59.5	-	61.3	0.9	-	0.9	4.6	-	4.7	0.097	-	0.077	0.061
Perdido Bay	14.0	-	14.5	0.1	-	0.1	0.6	-	0.6	0.047	-	0.044	0.034
Plum Island Sound	40.3	-	40.5	0.6	-	0.6	3.5	-	3.5	0.098	-	0.087	0.076

Table 4 Continued

	Area (km^2)			Stock to 0-15 cm (Tg)			Stock to 0-100 cm (Tg)			Density (g/cm^3)			Density (g/cm^3)
	Lo	-	Hi	Lo	-	Hi*	Lo	-	Hi*	0-15 cm	-	0-100 cm	At 100 cm
Puget Sound	37.5	-	46.0	0.3	-	0.3	0.9	-	1.1	0.046	-	0.025	0.019
Rio Grande	0.4	-	0.4	0.0	-	0.0	0.0	-	0.0	0.006	-	0.004	0.003
Rogue River	0.5	-	0.6	0.0	-	0.0	0.0	-	0.0	0.036	-	0.021	0.008
Rookery Bay	74.1	-	75.1	1.0	-	1.0	5.9	-	6.0	0.086	-	0.08	0.079
Sabine Lake	571.8	-	593.2	7.3	-	7.5	22.0	-	22.9	0.085	-	0.039	0.015
Saco Bay	17.0	-	17.9	0.1	-	0.1	1.0	-	1.0	0.052	-	0.057	0.076
San Antonio Bay	96.1	-	101.3	0.1	-	0.2	0.7	-	0.7	0.01	-	0.007	0.005
San Diego Bay	0.3	-	1.5	0.0	-	0.0	0.0	-	0.0	0.023	-	0.011	0.006
San Francisco Bay	157.7	-	212.0	1.9	-	2.5	10.3	-	13.8	0.08	-	0.065	0.057
San Pedro Bay	0.2	-	2.7	0.0	-	0.0	0.0	-	0.0	0.002	-	0.002	0.002
Santa Monica Bay	0.0	-	0.6	0.0	-	0.0	0.0	-	0.0	0.046	-	0.044	0.044

Table 4 Continued

	Area (km^2)			Stock to 0-15 cm (Tg)			Stock to 0-100 cm (Tg)			Density (g/cm^3)			Density (g/cm^3)
	Lo	-	Hi	Lo	-	Hi*	Lo	-	Hi*	0-15 cm	-	0-100 cm	At 100 cm
Sarasota Bay	9.3	-	10.2	0.1	-	0.1	0.3	-	0.3	0.07	-	0.027	0.014
Savannah River	259.5	-	271.6	3.0	-	3.1	16.1	-	16.8	0.076	-	0.062	0.055
Sheepscot Bay	7.6	-	9.0	0.1	-	0.1	0.6	-	0.7	0.086	-	0.073	0.068
Siletz Bay	1.8	-	2.1	0.0	-	0.0	0.1	-	0.1	0.044	-	0.037	0.035
Siuslaw River	5.6	-	5.9	0.0	-	0.0	0.3	-	0.3	0.052	-	0.05	0.031
South Ten Thousand Islands	0.0	-	893.8	0.0	-	0.0	0.0	-	0.0	0	-	0	0.000
St.Andrew/St.Simons Sounds	540.2	-	551.8	9.7	-	10.0	45.5	-	46.5	0.12	-	0.084	0.012
St.Andrew Bay	39.7	-	42.5	0.2	-	0.2	1.0	-	1.0	0.036	-	0.024	0.076

Table 4 Continued

	Area (km^2)			Stock to 0-15 cm (Tg)			Stock to 0-100 cm (Tg)			Density (g/cm^3)			Density (g/cm^3)
	Lo	-	Hi	Lo	-	Hi*	Lo	-	Hi*	0-15 cm	-	0-100 cm	At 100 cm
St.Catherines/Sapelo Sounds	522.4	-	532.3	8.4	-	8.6	49.5	-	50.4	0.108	-	0.095	0.087
St.Helena Sound	365.1	-	379.9	5.5	-	5.7	31.2	-	32.5	0.1	-	0.085	0.074
St.Johns River	73.1	-	78.7	1.0	-	1.1	3.9	-	4.2	0.096	-	0.053	0.017
St.Marys River/Cumberland Sound	182.4	-	184.7	3.0	-	3.0	17.1	-	17.3	0.11	-	0.094	0.089
Stono/North Edisto Rivers	170.7	-	177.8	0.1	-	0.1	0.5	-	0.5	0.004	-	0.003	0.002
Suwannee River	110.4	-	112.4	1.2	-	1.2	4.6	-	4.7	0.07	-	0.042	0.027
Tampa Bay	83.7	-	88.3	0.5	-	0.5	2.4	-	2.5	0.039	-	0.029	0.019

Table 4 Continued

	Area (km^2)			Stock to 0-15 cm (Tg)			Stock to 0-100 cm (Tg)			Density (g/cm^3)			Density (g/cm^3)
	Lo	-	Hi	Lo	-	Hi*	Lo	-	Hi*	0-15 cm	-	0-100 cm	At 100 cm
Terrebonne/Timbalier Bays	1029.3	-	1155.6	7.6	-	8.6	43.7	-	49.1	0.049	-	0.042	0.032
Tijuana Estuary	0.1	-	1.8	0.0	-	0.0	0.0	-	0.0	0.014	-	0.009	0.004
Tillamook Bay	4.6	-	5.0	0.0	-	0.0	0.2	-	0.3	0.054	-	0.052	0.051
Tomales Bay	0.6	-	4.1	0.0	-	0.0	0.0	-	0.1	0.024	-	0.013	0.008
Umpqua River	6.8	-	7.2	0.0	-	0.1	0.2	-	0.3	0.048	-	0.036	0.025
Upper Laguna Madre	8.7	-	9.7	0.0	-	0.0	0.0	-	0.0	0.006	-	0.004	0.003
Waquoit Bay	1.3	-	1.4	0.0	-	0.0	0.1	-	0.1	0.068	-	0.062	0.060
Wells Bay	3.3	-	3.5	0.0	-	0.0	0.1	-	0.1	0.042	-	0.04	0.039
West Mississippi Sound	728.3	-	780.3	7.9	-	8.5	48.9	-	52.4	0.073	-	0.067	0.053

Table 4 Continued

	Area (km^2)			Stock to 0-15 cm (Tg)			Stock to 0-100 cm (Tg)			Density (g/cm^3)			Density (g/cm^3)
	Lo	-	Hi	Lo	-	Hi*	Lo	-	Hi*	0-15 cm	-	0-100 cm	At 100 cm
Willapa Bay	37.8	-	44.5	0.3	-	0.3	0.9	-	1.1	0.051	-	0.025	0.015
Winyah Bay	335.0	-	339.9	3.2	-	3.3	19.8	-	20.1	0.064	-	0.059	0.054
Yaquina Bay	2.0	-	2.1	0.0	-	0.0	0.1	-	0.1	0.044	-	0.038	0.036
All CDAs	2000.7	-	2187.4	18.5	-	20.2	82.0	-	89.6	0.062	-	0.041	0.023

* Refer to table consistency section in text

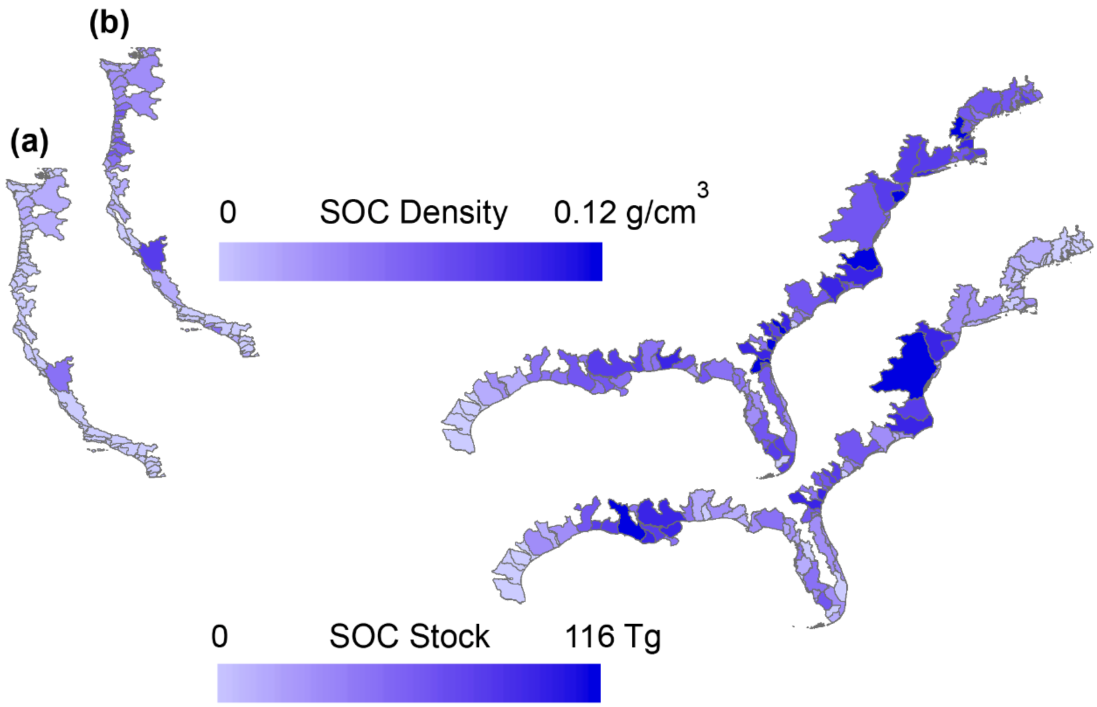

Figure 6. *(a) Total soil organic carbon across all tidal wetlands in each estuarine basin in the coastal United States (all estuarine basin names listed in Table 4), from 0-100 cm depth, with mapped values based on the minimum range (see text), (b) average SOC density in each estuarine basin, from 0-100 cm depth*

II.4 Discussion

For tidal wetlands within the continental United States, our results indicate that a total of 1,152.6 - 1,359.3 Tg of SOC are stored in the upper 100 cm of the soil profile (and 225.3 - 265.8 Tg in the upper 15 cm only), across a total of 24,945.9 km^2 of tidal wetland area. The SOCCR report estimated a total of 600 Tg C of carbon across 25,000 km^2 of estuarine wetlands in the conterminous United States (Bridgham *et al.*, 2007). This discrepancy can be attributed to the methods of evaluation of the estuarine wetlands carbon compilation. The portion of the SOCCR Report relevant to tidal wetlands, the

‘Estuarine Soil Carbon Pools’ data appeared to have been based on the analysis and literature compilation of Chmura *et al.,* 2003 which was skewed to near surface samples. For evaluation purposes, I compared the carbon values from our database versus Ouyang and Lee 2014 and Chmura *et al.,* 2003.

Based on the stocks database from the current study, I estimate carbon flux to be ~1.5 Tg C y^{-1} for tidal wetland burial in the continental US. This was obtained by assuming a carbon burial rate of 0.006 g cm^{-2} y^{-1}, and SOC density to be 0.03 g cm^{-2}. The latter was taken as a conservative estimate from Figure 3 for the upper 0-100 cm in an attempt to account for more recalcitrant carbon burial that is remaining after respiration and export to the water. Our flux estimates also assumed an average accretion rate of 0.2 cm yr^{-1} (Chmura *et al.* 2003; Callaway *et al.,* 2012; Morris *et al.,* 2016). Though I were unable to find a total flux value for the continental US from the literature to compare with, it has been estimated that there is ~220 Pg of carbon total and all wetlands (tidal and non-tidal) within the United States provide a carbon sink of ~49 Tg C y^{-1}, (Bridgham *et al.,* 2006). Globally, mangroves have been estimated to bury ~218 ± 72 Tg C y^{-1} (Bouillon *et al.*, 2008). Chmura *et al.*, (2003) and Duarte *et al.*, (2005a) have estimated global burial rates of 4.8 ± 0.5 Tg C y^{-1} and 87.2 ± 9.6 Tg C y^{-1} for salt marshes, respectively (McLeod *et al.*, 2011). Morris et al. (2012) estimated in the global coastal wetlands, there is a carbon burial of 4.5-15.8 Tg C y^{-1}. The spatial distribution of mangroves in certain regions are changing due to climate change. For example, more mature stands of mangroves are now found in more northern regions in the Gulf of Mexico due to a decrease in freezing events; this will have a significant impact on the type and amount of carbon stored and the overall vulnerability of blue carbon to storm events (Comeaux *et al*., 2011; Bianchi *et al*., 2012; Kulawardhana *et al*., 2015). According to Tampa Bay Blue Carbon Assessment, mangrove encroachment and expansion will be prominent as relative sea level rises, in many areas, to the detriment of other vital coastal wetlands such as salt marshes and coastal forests (Sheehan *et al.,* 2016). For mangroves, there are a multitude of other studies, giving a global carbon burial in Tg C y^{-1} of 31.1 ± 5.4 and 34.4 ± 5.9 (McLeod *et al.*, 2011). It is clear that further research is needed in order to accurately calculate the flux rate within the US tidal wetlands.

The area weighted average of the carbon densities was not greatly variant across the wetland types nor the coastal regions, yet given wetland type or coast, the standard deviations were typically vertical for the wetland types as shown in Figure 5, except for the emergent vegetation, which became less variable as depth increased. The coasts showed the highest variation; the Gulf Coast and the East Coast differing by nearly an entire degree of magnitude. Indeed, variation in carbon density at the individual polygon level is relatively high, and the finer the distinctions or categorizations that are drawn (Table 3, and then Table 4 with still finer categorizations), will show the greater the apparent spread of the reported values for average carbon densities across the categories (while the standard deviation of values within each category decreases). Thus, one must assess how the variation is spread within versus between the categories.

The area-weighted *average* density and the *total* carbon stock quantity for both the low and high estimates can be related through the wetland areas listed. However, there is an apparent discrepancy in the high estimate of the total carbon stocks when aggregated across all groups in a table (i.e. adding all rows down the high estimate column only) when compared to a similar aggregation from another table. If summed, the individual total high stocks for each category will result in a total high carbon stock that is similar though not exactly the same. The discrepancy occurs by using multiplication to increase the low estimate to the high estimate for each row in a table, based on the area of missing data throughout the original SSURGO dataset, and then adding these values across the rows. The only other option would be to fix the aggregation of the high total stocks to be exactly the same across all tables, but this would be artificial and not equal the sum of the individual rows. By allowing the discrepancies to be noticed, the individual categories with missing data can be recognized by the reader/user, and the call for further and more detailed information can be enforceable.

Still another complication is that for average weighted densities across a given depth range, any 'null' value found in that range for a specific horizon or component makes averaging invalid and that polygon cannot be used in the calculation, whereas the total carbon quantity can still be summed even when there are missing horizons or components. Because of these complications, it is important to note that all averages and totals reported here were calculated at the individual wetland polygon/record level, and hence short-hand math using values from portions of the summary tables may not reproduce the same values, unless one is able to investigate the source of the variation using the CoBluCarb database itself.

In estimating the carbon benefits of coastal wetland restoration or conservation projects, I suggest that the 0-15 cm depth values presented in this study be used by restoration project managers, as this is the typical rooting depth of newly-vegetated wetlands, but that for conservation projects on organic bearing mineral soils the greater depth of 0-100 cm is more relevant (Crooks *et al.,* 2009, Crooks *et al.*, 2011, Howard *et al.,* 2014). This depth is used as a default in many greenhouse gas inventories, such as the US National Greenhouse Gas Inventory and the IPCC default excavation values, developed in support of climate negotiations (Hiraishi *et al.,* 2014; Murray *et al.*, 2011). The 0-100 cm depth range can also be used as a first approximation for the depth of influence of released carbon from eroding coastal wetlands, from which much of the remobilized carbon will be returned to the atmosphere. In calculating the benefits of avoided emissions on drained organic soils, a different assumption is required. Here, the assumption is that emissions continue indefinitely, until either the entire carbon stock is exhausted or until water management changes to halt on going emissions (Emmer *et al.,* 2015). While a simplification, the 0-15 and 0-100 depth bins reflect the different zone of influence by human activities associated with restoration and wetland destruction. Restored or conserved wetland soil projects can yield investors a source of carbon market offset credits, or support a country's 'nationally determined contribution' to reduce emissions and meet goals of the Paris Climate Agreement.

There are opportunities for future improvements to this analysis. Within our study, the initial data from SSURGO and NWI are assumed to be accurate and based on uniform methods. As seen in our attempts to evaluate our dataset using previously published core data, the inconsistences in reporting among those efforts reveal the importance in utilizing a common dataset, developed using common methods. These inconsistencies can also be seen in some of the other properties and categorizations, such as percent of $CaCo^3$ and notation of mineral and organic soils. In the future, additionally, there needs to be more investigation into the validity of the van Bemmelen constant (0.58) for tidal wetlands as several studies have suggested that this canonical value may be too high (Craft *et al*., 1991; Callaway *et al.*, 2012; Keller *et al.*, 2015; Pribyl, 2010). Future work also could compare the use of NWI to that of the C-CAP (Coastal Change Analysis Program) database for the purpose of identifying tidal wetland areas. Factors such as latitude, temperature, slope, estuary type, among others, could be potential predictors of SOC density, when examined with the CoBluCarb database and could be helpful in determining priority areas for conservation and restoration. Finally, more data are needed for Monroe County, Florida, as this location contains a large percentage of the Everglades, specifically the South Ten Thousand Islands estuary, which likely contains the second-highest carbon quantity among all estuaries in US.

In summary, both restoration and conservation efforts can use the database and maps for the purpose of identifying locations that will maximize carbon accumulation and preservation. Academic and federal agencies can query the database to find relationships between soil organic carbon and other factors, such as sea-level rise or urban development. The demand for data products that can inform blue carbon investment and research is rising, and will continue to so in the context of both wetland conversion and rising atmospheric CO_2.

CHAPTER III

THE INFLUENCE OF ENVIRONMENTAL AND SOCIETAL FACTORS ON THE DISTRIBUTION OF TIDAL WETLAND SOIL ORGANIC CARBON IN THE CONTERMINOUS UNITED STATES

III.1 Introduction

Tidal wetlands comprise less than 1% of the total area within the conterminous United States, but their soils constitute 18% of the total carbon sequestration capacity (Bridgham et al., 2007). Globally, wetland soils are estimated to contain between 20% and 25% of total terrestrial carbon, and the sequestration per unit area is approximately 3 to 50 times greater than that of rainforests (Bridgham *et al.*, 2006; Nellemann *et al.*, 2009; Breithaupt *et al.*, 2012; Dodla *et al.*, 2012). Blue carbon, the soil carbon contained within these wetlands, can be a highly active sink for atmospheric CO_2 with potential economic value in the carbon markets (Howard *et al.*, 2014; McLeod *et al.*, 2011; Chmura *et al.*, 2003). Annually, blue carbon is credited with sequestering 0.9% to 2.6% of global anthropogenic CO_2 emissions, without compensating for the current loss rate of global tidal wetlands (Murray *et al.*, 2011).

Soil organic carbon (SOC) in tidal wetlands is mostly attributed to sediment burial from sea level rise (Chmura *et al.*, 2003; Bridgham *et al.*, 2007), however there is a balance between inputs from the sediments and vegetation with outputs from CO_2 efflux, and leaching to water and recalcitrant soil aggregates (De Deyn *et al.*, 2008; Davidson and Janssens, 2006; Kirwan and Blum, 2011). These ecosystems have high organic matter inputs, slow mineralization and decomposition rates, and high vertical accretion rates creating a large sink for atmospheric carbon (Dodla *et al.,* 2012; Mcleod *et al.,* 2011). While non-tidal wetlands such as peatlands are also considered large potential carbon sinks, saltwater intrusion within tidal wetlands causes sulfate reducers to become relatively more important than methanogens since they typically thrive in an anaerobic soil environment (Whiting and Chanton, 2000). As a result, tidal wetlands produce relatively small quantities of methane, a greenhouse gas with potency 25 times that of CO_2 on a molar basis (Whiting and Chanton, 2000). Even though there are massive amount of SOC in all wetlands, tidally influenced wetlands are particularly important due to this lack of methane outputs, their potential as CO_2 sinks, and negligible N_2O emissions (Chmura *et al.,* 2003; David and Jenssens, 2006).

Long term soil organic carbon burial rates have been attributed to changes in physical (temperature, nutrients, sea-level, disturbance, etc.) and biological (vegetation competition and speciation, nutrient loading, etc) parameters (Mcleod *et al.,* 2011; Morris and Bradley, 1999). Especially within anaerobic soils, decomposition in the soil can be limited based on the temperature sensitivities and affect the amount of CO_2 being respired back to the atmosphere (Davidson and Janssens, 2006; Raich and Schlesinger, 1992; Kirwan and Blum, 2011). A warming and changing climate can drive sea level rise and alter other environmental thresholds such as precipitation, resulting in an indirect human influence on wetland loss and encroachment (Couto *et al.,* 2013; Pachauri and Reisinger 2007; Bianchi *et al.,* 2012). Besides changes in climate, human impacts such as the drainage or partial drying of wetlands can transform an ecosystem into a net source or CO_2 and CH_4 instead of a sink (Burkett and Kusler, 2000). Fragmentation or loss of a wetland can cause basic processes to fail and lead to a decrease in biodiversity in a weakened environment (Cline *et al.*, 2007).

Between the years 2004 and 2009, there was an estimated 1.4% loss of coastal wetland area, with human-induced climate change being the greatest cause of loss, particularly along the coast of the Gulf of Mexico (Dahl, 2009). Agricultural land conversion, from the mid-1950s to the mid-1970s, was the primary reason for direct total wetland destruction, and accounted for an estimated annual net loss of about 455, 000 acres (Brown and Lant, 1999). Non-agriculture land conversion uses peaked between the years 1974-1983 with an estimated 185,700 acres lost per year (Brown and Lant, 1999).

Throughout much climate change literature, there is a call for evaluation of the variables and controls on soil organic carbon, though most of these studies focus on terrestrial ecosystems evaluations (Lal, 2010; Powlson *et al.*, 2011). While many variables have been noted, such as temperature, precipitation, limiting nutrients, the degree to which these variables impact, therefore influence, the rate the density of SOC in a variety of other ecosystems, such as coastal wetlands are unknown.

The purpose of this study is to determine the environmental conditions that could be related and potentially influential to the sequestration of soil organic carbon in coastal USA. There are three main motivations that drive this study:

1) Ecosystem-level analysis could provide a better understanding for the conditions that lead to enhanced or degraded carbon sequestration rates in times of rapid global change

2) Large scale ecosystem characteristics, such as temperature, precipitation, longitude, etc. could reveal if carbon sequestration is influenced more by terrestrial or oceanic conditions
3) An understanding of the variables that drive the unique histories and aspects of coastal wetlands in United States' estuaries, such as human influence, typology, and vegetation type, could lead to future conservation efforts within specific estuarine boundaries

III.2 Methods

In this study, I examined soil organic carbon density in conjunction with associated physical and environmental characteristics at three spatial scales: individual wetland (local), estuarine basin (regional), and national scale. Categorical factors, such as typology, vegetation type, and human influences, were analyzed by ANOVA and then Tukey post hoc analysis on a local scale. Based on significant differences between levels of wetland SOC found during the local scale analysis, I then developed regression models accordingly for the regional scale. Continuous numerical variables were analyzed by univariate linear regression, as well as with multivariate regression.

Soil organic carbon densities were considered on a 0-15 cm soil column basis, an approximate depth interval strongly influenced by productive root communities and surface influences. Wetland soils greatly vary spatially, both horizontally and vertically, and generalizing assumptions would have been more difficult to assume at greater depths in the soil column. SOC densities were aggregated separately for each scale from the United States soil organic carbon database, CoBluCarb (Hinson *et al.,* 2017). This database was created using spatially-explicit data from United States Department of Agriculture's Soil Survey Database (SSURGO) and National Wetland Inventory (NWI) database. In Hinson et al. (2017), all methods and assumptions for CoBluCarb are discussed. To our knowledge, CoBluCarb is the most accurate and current consolidation of wetland soil organic carbon values for the United States.

III.2.1 *Local Analysis (ANOVAs)*

I initially considered five different aspects that could potentially influence soil organic carbon: location (represented by coasts due to the latitude and longitude gradients), estuarine typology, vegetation type, water regime, and management regime. Comparisons within these factors were made on an individual wetland basis. In total, there were 605,276 tidal wetland polygons in the United States that I considered. Each of these wetland polygons were considered as an individual location. The SOC was considered to represent the total organic carbon within the soil and was not weighted in any capacity. The location was determined by natural environmental and political divisions around the US. The East and West Coasts are both arrayed more across latitude, whereas the Gulf Coast arrayed more across longitude.

Based on the estuarine typology descriptions described in Bianchi (2007), there were six topological levels considered: former river valley primary estuary, former glacier valley primary estuary, river delta primary estuary, tectonic structural primary estuary, coastal lagoon secondary estuary, and unclassified estuary. These six levels were grouped in primary and secondary estuary systems (Fig. 7).

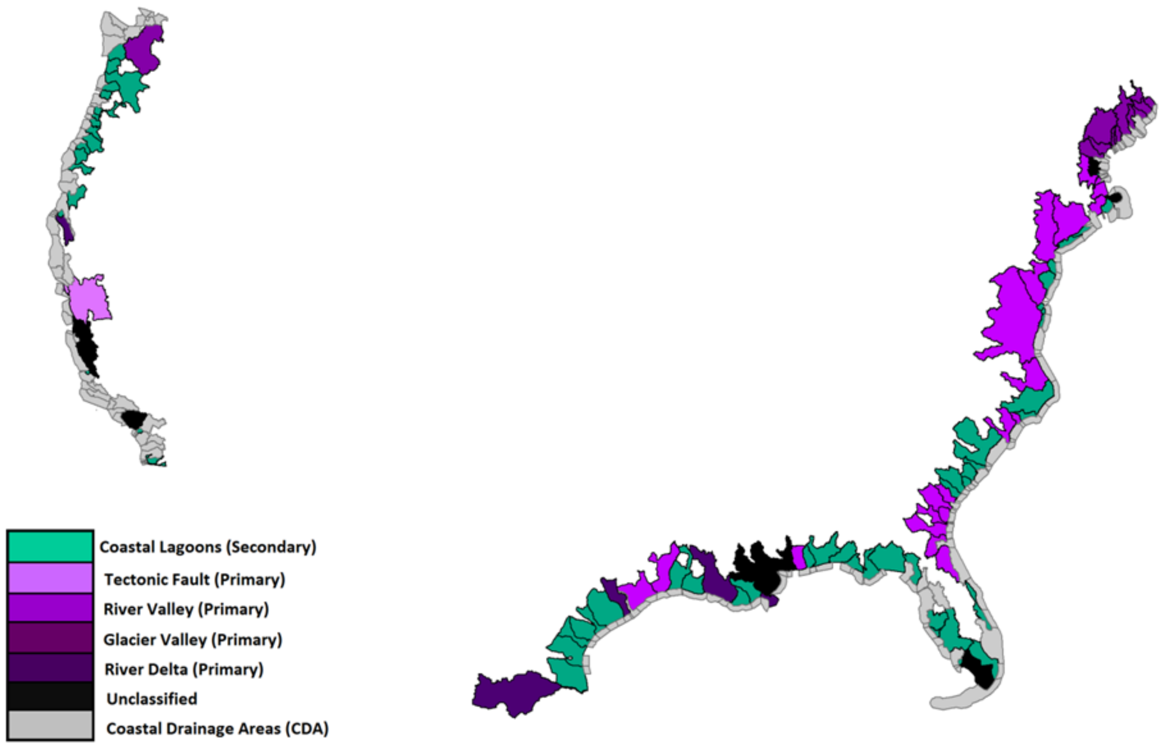

Figure 7. *Primary and secondary estuaries in the United States, including primary estuary sub-typologies*

The Coastal Assessment Framework (National Ocean Service) defines both estuarine (EDAs) and coastal drainage areas (CDAs). The majority of tidal wetlands fall within the EDAs and the Bianchi (2007) typology is an appropriate method to group them. However, a relatively small total area of tidal wetlands within the US fall within the CDA boundaries, though there can be large spatial gaps in the coverage across the US if these basins were not considered. Thus, these CDAs were considered within the category of unclassified estuaries.

There were four different vegetation levels considered in this analysis, based on the CoBluCarb classification scheme, which is based on NWI wetland levels: estuarine emergent vegetation, tidal freshwater wetlands, estuarine shrub scrub vegetation wetlands, and estuarine forested wetlands. In this study and the Cowardin classification system (Cowardin, 1979), they are as follows: (1) estuarine emergent (rooted graminoid and herbaceous hydrophytes that are present through the year, typically salt marshes and brackish marshes), (2) estuarine shrub scrub (woody vegetation that is less than 6 meters high, typically small black and red mangroves in the southern United States and woody shrubs in Northern United States), (3) estuarine forested (more swamp-like wetlands, including woody species that are greater than 6 m in height, usually taller red mangroves in the southern US and taller non-mangrove tree species in the northern United States), and (4) freshwater tidal (all marsh, shrub, and forested wetlands , palustrine and riverine, that are tidally influenced, but with salinities less than 0.5 ppt).

The water regime levels were derived separately for freshwater and saltwater proportions, by the codes listed in NWI wetland classification system (Cowardin, 1979). For both freshwater and saltwater, there were only two different regimes considered within each. There were more regimes within the Cowardin wetland classification system, but only four were considered due to the lack of unique identification (re-usage of letters in the NWI coding system) and lack of water regime notations in all wetlands considered. For example, the code “R” is used to denote a riverine wetland as well as a seasonally flooded freshwater tidal wetland. These codes are case-sensitive and the order varies among each individual wetland. Due to the inconsistency of order as well as the repetition, these were grouped in the "Unknown" category to avoid double counting locations. The two saltwater water regimes were subtidal and regularly flooded which are defined as permanently flooded with tidal water and alternating flooding and land exposure at least once a day respectively. The two freshwater regimes were semi-permanently flooded and permanently flooded. Semi-permanently flooded is defined as tidally influenced surface water present for most of the year and permanently flooded is defined the same as subtidal in the saltwater water regime.

The management regime levels were determined from the NWI classification system as wetlands that are natural, have had human modifications, or human-constructed wetlands. The levels of modifications defined in this study include partially drained or ditched (wetlands in which the water has been artificially lowered but still supports hydrophyte vegetation), diked or impounded (a human made barrier or dike has been placed to obstruct water inflow or outflow), artificial (an artificial or natural substrate placed by humans to create a wetland environment), spoil (spoil material has been placed to enhance the environment), and excavated (located in a channel or basin excavated by humans) (Cowardin, 1979). There were no farmed wetlands in this study. There were other factors initially considered such as soils (mineral or organic) and water chemistry, but eventually rejected these based on too few data points for proper statistical analysis, as well as overlap with other tested factors.

III.2.2 *Regional and National Analysis (Regression Models)*

There was a total of 45 factors considered at regional and national scale, with each individually regressed against the area-weighted soil organic carbon density. These 45 factors were derived from a range of environmental aspects that could influence SOC density, including geographical, oceanic, terrestrial, societal, and atmospheric differentiations. Many of these variables originated and were calculated from the National Estuarine Eutrophication Assessment (NEEA), National Oceanic and Atmospheric Administration's Coastal Assessment Framework (CAF), and the United States Geographical Survey's SPAtially Referenced Regressions On Watershed attributes (SPARROW). The geographical variables include both catchment (total watershed basin) and estuary (estuarine waters only) latitude and longitude values.

To accomplish this, I first summarized the individual wetland polygon data within each estuary (dependent factor). Each independent factor was also derived within the spatial boundaries of the 115 estuarine basins from the CAF dataset. Statistically significant factors from the ANOVA analysis that could logically be analyzed on a regional basis were considered in regression analysis along with the national comparisons of the factors. Initial models tested for linear univariate relationships (factor as it related to SOC, on an estuary by estuary basis). Statistically significant findings were then considered further multivariate regression models.

As based on the ANOVA analysis described above, the levels that were significantly different among one another within a given category were then also considered on the regional basis as well. Regional basis in this study is defined synonymously as ‘estuary by estuary’, or on the estuarine scale considered across regional groupings (such as the West Coast). For example, estuarine typologies could be further categorized on primary and secondary as two distinct groups, and separate regression models could be created for the estuaries falling into each, but entire estuarine basins cannot be categorized as one specific management regime or water regime due to the various levels and complexities of tidal wetlands within each estuary. The temperature and precipitation values considered were 30 year means for each estuarine basin (Oregon State PRISM, 2017).

III.3 Results

III.3.1 *Local Analysis (ANOVAs)*

Of the five factors considered with ANOVA analysis at the local scale, there were two that allowed consolidation on a further regional basis. These factors, location and typology, contained the most statistically significant differences among their constituents (Fig 8). All of the coasts and vegetation levels were significantly different from one another with extremely low p values of 0.000-0.001.

For the primary sub-typologies, river deltas and tectonic faults were significantly different from all other typologies. Coastal lagoons (secondary estuary) and coastal drainage areas (CDA) are not significantly different, nor is the glacier and river valleys significantly different.

For the water regimes, subtidal, permanently flooded, and regularly flooded were not significantly different from one another. Permanently and semi-permanently flooded, both freshwater tidal regimes, were not significantly different from one another, nor from the subtidal saltwater category. Permanently flooded, subtidal, and unclassified were also not significantly different from one another.

Management regimes varied greatly across the coastal United States. Artificial was not significantly different from any other management regime, including natural, though the only other regimes that were not significantly different were spoil, artificial and natural together. All other management regimes SIC are significantly different from one another.

The local analyses were all one-way ANOVAs with p-values on a 95% confidence level as determinants for significance. All significant factors were analyzed using a Tukey post hoc analysis to determine which levels were significant in reference with each other.

III.3.2 *Regional and National Analysis (Regression Models)*

At a 95% confidence level, the groups of the West and the Gulf Coast estuaries each had several significant correlations (corr) between various variables and SOC. The East Coast only had a few significant correlations, even when considering the results at a 90% confidence level (Table 5). In the following descriptions, only those correlations with p values < 0.05 are described.

There were significant relationships based on location. Overall, besides the geographical variables such as longitude and latitude, the oceanic variables were the most influential on the West Coast. Sea surface temperature (r^2 of 0.49; corr: neg), oceanic salinity (r^2 of 0.38; corr: neg), oceanic nitrate (r^2 of 0.42; corr: pos), and oceanic dissolved inorganic phosphorous (DIP) (r^2 of 0.47; corr: pos) were all relatively high.

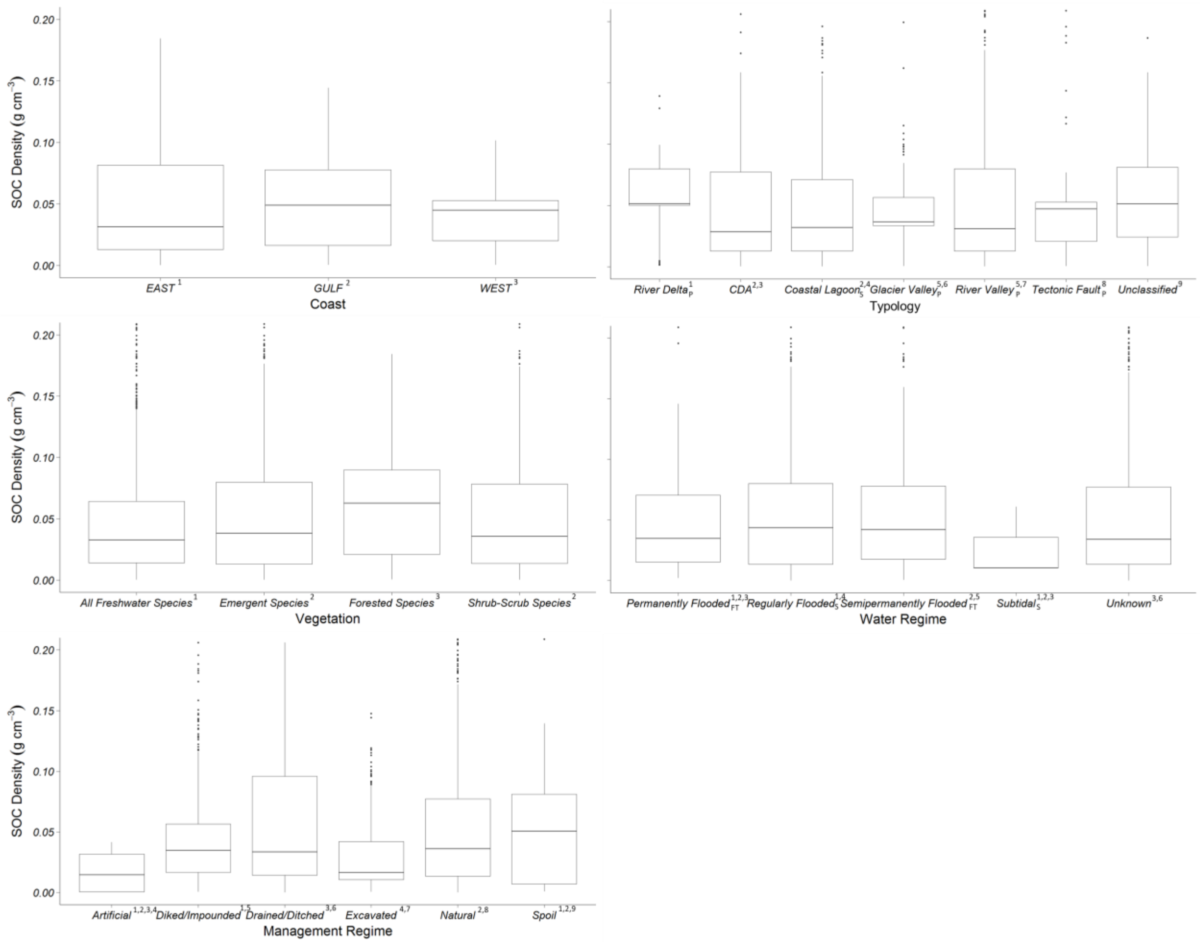

Figure 8. *Categorical results for all 5 factors of tidal wetlands. Results of the Tukeys post hoc analysis are represented by numbers and larger factors such as typology and salinities are also listed. P represents primary estuaries and S represents secondary estuaries. FT represents freshwater tidal and S represents saltwater tidal wetlands.*

Table 5. Single variable linear regressions based on location and typology of estuaries in the United States then as the total regression based on all estuaries in the United States. Regressions are all single variables or the total variables regressed again soil organic carbon density. Dark grey shaded boxes denote significant relationships based on at the p < 0.05 level. Light grey shaded boxes denote those relationship significant within at the p < 0.10 level.

	West		Gulf		East		Primary Estuary		Secondary Estuary		United States	
	r^2	p	r^2	p	r^2	p	r^2	p	r^2	p	r^2	p
Agricultural Area	0.030	0.365	0.000	0.969	0.000	0.878	0.009	0.516	0.008	0.482	0.001	0.770
Air Temperature (A)	0.422	0.000	0.035	0.296	0.027	0.241	0.001	0.865	0.004	0.609	0.007	0.367
Barren Area	0.039	0.305	0.000	0.983	NA	NA	0.007	0.560	0.019	0.276	0.001	0.731
Catchment Elevation(A)	0.000	0.919	0.042	0.252	0.001	0.835	0.131	0.009	0.121	0.005	0.110	0.000
Catchment Elevation(Max)	0.009	0.624	0.018	0.463	0.005	0.608	0.097	0.026	0.117	0.006	0.075	0.003
Catchment Latitude	0.496	0.000	0.023	0.403	0.022	0.295	0.001	0.803	0.007	0.518	0.009	0.317

Table 5 Continued

	West		Gulf		East		Primary Estuary		Secondary Estuary		United States	
	r^2	p	r^2	p	r^2	p	r^2	p	r^2	p	r^2	p
Catchment Longitude	0.418	0.000	0.225	0.005	0.029	0.227	0.315	0.000	0.251	0.000	0.316	0.000
Drainage Area	0.002	0.798	0.001	0.870	0.002	0.740	0.013	0.422	0.013	0.372	0.003	0.560
Estuary Depth	0.004	0.732	0.005	0.696	0.015	0.384	0.084	0.039	0.002	0.754	0.005	0.448
Estuary Latitude	0.480	0.000	0.058	0.175	0.020	0.317	0.003	0.710	0.006	0.538	0.008	0.345
Estuary Longitude	0.469	0.000	0.230	0.005	0.028	0.239	0.317	0.000	0.243	0.000	0.313	0.000
Estuary Perimeter	0.073	0.158	0.029	0.341	0.003	0.711	0.004	0.668	0.006	0.531	0.013	0.235
Estuary Salinity (A)	0.126	0.059	0.029	0.347	0.060	0.080	0.022	0.294	0.006	0.553	0.013	0.228
Estuary Volume	0.003	0.770	0.011	0.556	0.006	0.596	0.017	0.360	0.047	0.087	0.000	0.903

Table 5 Continued

	West		Gulf		East		Primary Estuary		Secondary Estuary		United States	
	r^2	p	r^2	p	r^2	p	r^2	p	r^2	p	r^2	p
Evaporation[(D)]	0.071	0.162	0.002	0.804	0.006	0.601	0.001	0.790	0.007	0.508	0.011	0.277
Flushing Time	0.013	0.550	0.011	0.569	0.008	0.529	0.119	0.013	0.003	0.673	0.026	0.089
Forest Area	0.025	0.417	0.000	0.931	0.002	0.767	0.005	0.636	0.007	0.520	0.000	0.841
Freshwater Inflow [(A)(D)]	0.002	0.801	0.001	0.861	0.001	0.868	0.009	0.506	0.006	0.557	0.001	0.696
Frost Days per Year	0.406	0.000	0.050	0.210	0.031	0.212	0.034	0.195	0.059	0.055	0.067	0.006
Mixed Zone Area	0.186	0.019	0.007	0.649	0.003	0.682	0.000	0.881	0.026	0.208	0.011	0.268
Non-tidal Wetland Area	0.109	0.081	0.009	0.598	0.060	0.081	0.001	0.835	0.131	0.004	0.010	0.285
Ocean DIP	0.471	0.000	0.012	0.539	0.021	0.309	0.043	0.147	0.008	0.484	0.011	0.269

Table 5 Continued

	West		Gulf		East		Primary Estuary		Secondary Estuary		United States	
	r^2	p	r^2	p	r^2	p	r^2	p	r^2	p	r^2	p
Ocean NO_3	0.419	0.000	0.180	0.014	0.029	0.224	0.000	0.928	0.073	0.032	0.040	0.032
Ocean Salinity $^{(A)}$	0.382	0.000	0.008	0.620	0.007	0.560	0.017	0.367	0.004	0.629	0.005	0.466
Ocean Salinity $^{(Max)}$	0.393	0.000	0.077	0.118	0.008	0.525	0.011	0.471	0.003	0.691	0.002	0.628
Ocean Salinity $^{(Min)}$	0.381	0.000	0.035	0.294	0.006	0.600	0.026	0.258	0.032	0.159	0.036	0.042
Open water Area	0.066	0.179	0.010	0.577	0.005	0.612	0.005	0.628	0.017	0.307	0.017	0.161
Population	0.006	0.581	0.000	0.933	0.002	0.841	0.009	0.507	0.003	0.660	0.000	0.941
Precipitation $^{(A-30yr)}$	0.359	0.001	0.500	0.000	0.002	0.784	0.090	0.031	0.073	0.032	0.040	0.033

Table 5 Continued

	West		Gulf		East		Primary Estuary		Secondary Estuary		United States	
	r^2	p	r^2	p	r^2	p	r^2	p	r^2	p	r^2	p
Precipitation[(D)]	0.049	0.247	0.022	0.413	0.007	0.546	0.002	0.728	0.037	0.134	0.022	0.117
Rangeland Area	0.003	0.779	0.001	0.837	0.002	0.741	0.010	0.491	0.032	0.161	0.005	0.472
Residence Time	0.002	0.815	0.000	0.983	0.004	0.668	0.030	0.223	0.062	0.049	0.004	0.519
Riverine Nitrogen Flux [(A)(D)]	0.009	0.638	0.000	0.916	0.001	0.844	0.012	0.445	0.003	0.669	0.000	0.836
Riverine Organic Carbon Flux [(A)(D)]	0.002	0.840	0.002	0.800	0.004	0.656	0.012	0.449	0.001	0.808	0.000	0.969
Riverine Organic Carbon Flux[(A)(D)(S)]	0.003	0.776	0.005	0.697	0.017	0.369	0.007	0.557	0.014	0.355	0.001	0.740

Table 5 Continued

	West		Gulf		East		Primary Estuary		Secondary Estuary		United States	
	r^2	p	r^2	p	r^2	p	r^2	p	r^2	p	r^2	p
Riverine Phosphorus Flux	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Saltwater Zone Area	0.026	0.401	0.003	0.758	0.002	0.782	0.003	0.717	0.002	0.740	0.006	0.431
Sea Surface Temperature(A)	0.493	0.000	0.002	0.800	0.014	0.398	0.003	0.697	0.000	0.969	0.000	0.850
Temperature(A-30yr)	0.380	0.000	0.106	0.061	0.023	0.284	0.017	0.364	0.008	0.478	0.021	0.126
Tidal Flow	0.047	0.258	0.037	0.283	0.009	0.497	0.011	0.471	0.076	0.029	0.040	0.034
Tidal Fresh Zone Area	0.011	0.582	0.022	0.413	0.020	0.319	0.013	0.434	0.001	0.791	0.016	0.182

Table 5 Continued

	West		Gulf		East		Primary Estuary		Secondary Estuary		United States	
	r^2	p	r^2	p	r^2	p	r^2	p	r^2	p	r^2	p
Tidal Height	0.328	0.001	0.103	0.069	0.001	0.827	0.043	0.145	0.000	0.882	0.043	0.027
Tidal Volume	0.047	0.258	0.019	0.441	0.009	0.497	0.009	0.518	0.056	0.063	0.036	0.042
Tidal Wetland Area	0.208	0.013	0.050	0.204	0.035	0.182	0.002	0.753	0.090	0.017	0.030	0.062
Tidal Wetland Area$^{(S)}$	0.253	0.006	0.091	0.093	0.026	0.258	0.019	0.339	0.045	0.094	0.034	0.053
Urban Area	0.048	0.253	0.000	0.953	0.003	0.686	0.005	0.610	0.003	0.667	0.000	0.976
Wind Speed	0.223	0.010	0.442	0.000	0.030	0.218	0.015	0.396	0.012	0.397	0.028	0.077

$^{(A)}$ is average or mean calculations

$^{(D)}$ is daily calculations

$^{(Max)}$ and $^{(Min)}$ is maximum and minimum calculations

$^{(S)}$ is calculations from SPARROW

Location was a major factor on the West Coast, both catchment latitude (r^2 of 0.5; corr: pos) and longitude correlations (r^2 of 0.41; corr: neg) and estuary latitude (r^2 of 0.48; corr: pos) and longitude (r^2 of 0.47; corr: neg). There were other significant physical variables, including air temperature, wind speed, frost days per year, tidal height, mixed zone area, and total tidal wetland area. Precipitation (r^2 of 0.36; corr: pos) and temperature (r^2 of 0.38; corr: neg) were also significant. Temperature and precipitation are typically assumed to be the most important variables for wetland and SOC deposition though only precipitation was also highly significant on the Gulf Coast (r^2 of 0.5; corr: pos) (Fig 9). Longitude was the only significant geographical factor on the Gulf Coast with a catchment and estuary longitude (r^2 of 0.23; corr: pos). On a 95% confidence level, the only other physical variables significant on the Gulf Coast were oceanic nitrate (r^2 of 0.18; corr: pos) and wind speed (r^2 of 0.44; corr: neg). No variables were significant on the East Coast within a 95% confidence level. When all the significant variables were addressed in a multiple regression, the Gulf Coast r^2 was increased to 0.55 and the West Coast increased to 0.87.

Primary and secondary estuaries varied from each other but the correlations were generally very low. For both primary and secondary estuaries respectively, elevation (r^2 of 0.13 and 0.12; corr: neg), longitude (r^2 of 0.32 and 0.25; corr: pos), and precipitation (r^2 of 0.09 and 0.07; corr: pos) were significant. Secondary estuaries' SOC also had significant correlations with non-tidal (r^2 of 0.13; corr: pos) and tidal wetland area (r^2 of 0.09; corr: pos), oceanic nitrate (r^2 of 0.07; corr: pos), tidal flow (r^2 of 0.08; corr: pos), and residence time (r^2 of 0.06; corr: pos) while primary estuaries SOC was only related to estuarine depth (r^2 of 0.08; corr: neg) and flushing time (r^2 of 0.12; corr: neg).

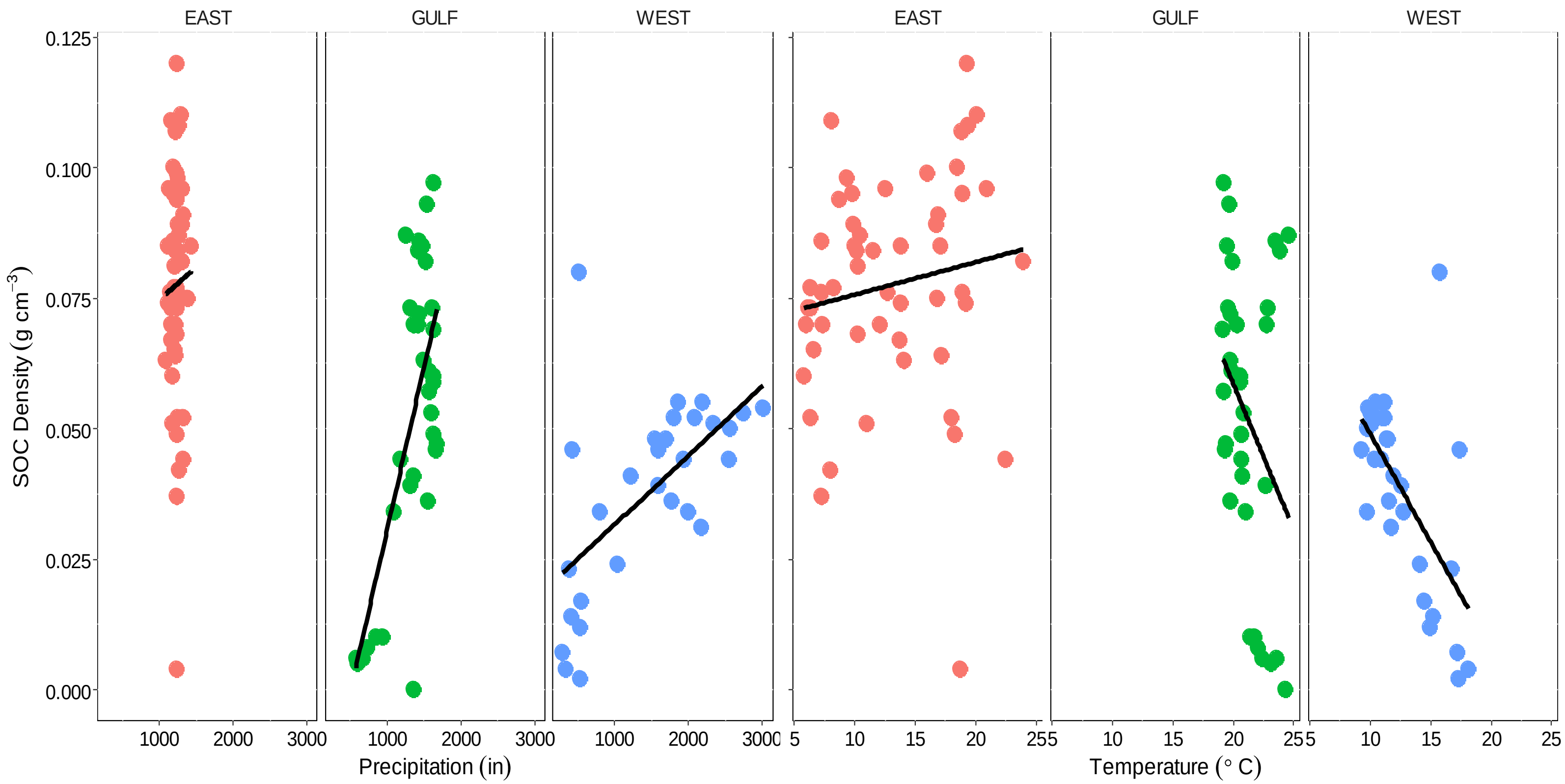

Figure 9. *Tidal wetland soil organic carbon density as a function of temperature and precipitation*

III.3.3 *National Totals and Analysis*

Longitude (r^2 of 0.31) was the greatest factor SOC density on the national scale. There were relationships, but the correlation is typically low. There were correlations between SOC and elevation (r^2 of 0.11), frost days per year (r^2 of 0.07), oceanic DIP (r^2 of 0.01), oceanic nitrate (r^2 of 0.04), minimum oceanic salinity (r^2 of 0.04), precipitation (r^2 of 0.04), tidal flow (r^2 of 0.04), tidal height (r^2 of 0.04), and tidal volume (r^2 of 0.04). Longitude was the only substantial variable, leading to the assumption that it would be better to look at the relationships on a smaller scale that highlights the differences between longitudes. Overall, when all significant variables were considered in a multivariate model, the r^2 was increased to 0.52.

III.4 Discussion

Relationships between environmental conditions and ecosystem processes can change drastically depending on the scale of the analysis. Many times, coarse scale relationships will be less obvious or more obvious on a finer, more detailed scale. Nationally, there are many conditions that appear to be related, potentially even strongly related, to the overall wetlands SOC density for the estuaries. Some of the geographical variation, such as longitude, was minimized when the variables are considered regionally, allowing more detailed environmental condition's influences to be observed. The only two coasts with significant relationships for $p < 0.05$ were the West and the Gulf Coasts. Further research will be needed to determine what environmental conditions are most predictive for the East Coast, if any.

Many studies use temperature and precipitation as key indicators for soil organic carbon controls and soil respiration in ecosystems (Raich and Schlesinger, 1992; Whiting and Chanton, 2000). Based on our findings, using only precipitation and temperature as indicators for soil organic carbon misses the complexity. There is a large diversity of wetlands and the predictive conditions for SOC are difficult to summarize on the coarsest scales, especially as compared with other ecosystems (Drexler *et al.*, 2004). Precipitation was better correlated that temperature was with SOC, on every scale, especially on the Gulf Coast where the growth of many ecosystems is dependent on the rainfall amount, but curiously precipitation was not significantly related to SOC on the East Coast. Nationally, precipitation has low correlation with no clear trends, but the variety among the coasts and wetlands on such a large scale create an understated appearance as opposed to the significant impact of precipitation on the Gulf and West Coasts. Future models should incorporate other key variables besides precipitation and temperature to obtain accurate soil organic carbon conditions.

The West Coast of the United States has the greatest amount and highest correlation with the environmental variables considered in this study. Many of the greatest correlations were between the oceanic variables and West Coast soil organic carbon. There are many possible reasons that the West Coast soil organic carbon could be more interactive with oceanic variables than the other coasts. A major difference between the West coast and the other United States' coasts are the temperature of the oceanic currents. The California Current on the Pacific is a cold stream current, while the Gulf Stream (both Gulf and East Coasts) is a warm stream current. The temperature of the water can affect decomposition rates as well as erosion through plant growth and animal activity. Decomposition rates decrease in colder temperatures, allowing for less breakdown and accumulation of peat in wetlands. Due to this, it would be assumed that the differences in temperatures could affect the nutrient input into estuaries with high oceanic inputs.

The West Coast of the United States is a known upwelling zone, causing abundant nutrients that affect fisheries, productivity, and nutrient availability to coastal estuaries. Upwelling zones are areas where wind driven currents push the nutrient depleted surface water away from the coast, allowing deeper colder nutrient rich waters to rise to the surface (Zatsev *et al.*, 2003). These zones can bring up many limiting nutrients to the surface waters, including nitrate, silicate, iron, and phosphate. Many times, upwelling current can more intense on geological boundaries. This type of zone on the coast could indicate the correlation for wind speed and soil organic carbon on the coasts. As a tectonic boundary also, the geological boundary may be intensifying the upwelling currents, strengthening the correlation between oceanic nitrate and soil organic carbon densities on the West Coast.

The tectonic boundary can also provide difference between the bathymetry and typology of the estuaries. On the West Coasts, many of the primary topological divisions are sub typologies that are sporadic and rare on the other Coasts. The topological differences in among the estuaries result is differences between the SOC densities, though while interesting, would need further investigation to a major variable in SOC deposition. Geographical variables such as coasts, and consequently latitude and longitude, are more qualified as predictors of SOC densities than typology.

While current strength and bathymetry were not tested in this study and may have an effect, the estuarine tidal height can be based on the typology of the estuary, specifically the size of the opening of the estuary. Tidal height can cause vertical accretion and erosion, depending on the longshore drift, force, and frequency of the tides. In Guo et al. (2009), continuous carbon flux in an estuarine wetland changed drastically depending on the level and duration of inundation of the wetland soils. This supports our finding that tidal influences have a significant effect on potential carbon sequestration.

Many of the significant variables discovered in this study will be modified as a result of changing climate, resulting in altered sequestration rates and stock quantities. Coastal wetlands have the ability to adjust and migrate to cope with the results of climate change such as sea level rise, storm intensity and frequency, increased precipitation, and ocean acidification, but at what rate or duration is unknown. Currently there is not enough research to understand if changing environmental variables will help or hurt the carbon sequestration in these tidal wetlands. According to Day et al. (2008), river deltas could be experience some of the highest relative sea level rise, due to the compaction and dewatering of sediments, as well as extraction of oil and natural gas by humans. In these areas, such as the Mississippi delta, mangroves could possibly be drowned by the high levels of sea level rise, but studies have shown that many times, tidal wetlands can match the critical sea level rise, creating uncertainty on how these important environments can react and how the carbon will be affected. Many of these climate changes can intensify or counteract the effects on wetlands. Conservation of key components of wetlands will be important to mitigate the damage to the soil organic carbon levels.

III.5 Conclusions

Singularly, environmental parameters did not have as strong of a correlation with SOC density compared to multiple environmental variables. However, there are still variables that are not accounted for that make up large portions of the relationships with SOC density in tidal wetlands. There are multiple different levels and possible relationships between the environment and SOC density. These can be examined in levels or variables on multiple different scales. More research will be needed to create more accurate variables and a greater variety of variables for understanding the role of other aspects in the environment in SOC density. Overall, regional divisions can give important insight into the variables that predict and affect the soil organic carbon density. These calculations and relationships are important to understand as ecosystems and society adapt to changing climates. Understanding how and how much these environmental variables affect and influence tidal wetland SOC can provide insights for management in the changing environment and climates.

CHAPTER IV

FATES AND FLUXES OF THE CONTERMINOUS UNITED STATES' TIDAL WETLAND SOIL ORGANIC CARBON

IV. 1 Introduction

Tidal wetlands are among the most productive ecosystems in the world, but comprise only about 1% or less of the world's landmass (Howard et al., 2017; Pidgeon 2009). These ecosystems are natural carbon sinks, with an estimated 1,153-1,359 Tg of SOC in the upper 0-100 cm of soils within the United States alone and have burial rates 6 – 10 times greater than adjacent upland ecosystems (Hinson et al., 2017; Pidgeon, 2009). Tidal wetlands are unique ecosystems, not fully categorized as terrestrial or marine, that allow large carbon deposits to be stored in sediments similar to marine environments, but with the above ground biomass of terrestrial environments (Pidgeon, 2009). Up to 98% of the total carbon pool is sequestered in the soil as compared to the living biomass (Donato et al., 2011; Bouillon et al., 2008). The carbon sequestration potential of these wetlands is estimated to be as much as 3 to 50 times greater than that of other terrestrial ecosystems, including rainforests, per unit area (Howard *et al.,* 2017; Mcleod *et al.,* 2011; Chambers *et al.,* 2011; Pidgeon, 2009). Tidal wetland carbon is not only influential within the boundaries of the wetlands, but the export to the surrounding estuaries as well through lateral fluxes in these hydric, typically organic soils (Cai, 2011).

The Outwelling Hypothesis, the assumption that wetlands are a net source of nutrients and organic matter to the surrounding water and environments, was the accepted theory on energy and nutrient flow through wetlands and the surrounding environments (Odum, 1968; Teal 1962; Nixon, 1980). While this was the accepted theory for decades, there were few studies directly challenging the theory and those that did were typically limited to the East Coast and salt marshes (Lee, 1995; Taylor and Allanson, 1995; Childers *et al.*, 2002). However, studies that did challenge the theory discovered that the conclusions were variable, and whether a wetland was a net-exporter depended on the physical and chemical properties of the wetland (Childers *et al.*, 2002). Location, such as high marshes or low marshes, have also been used as examples for the prominence of the Outwelling Hypothesis (Taylor and Allanson, 1995; Childers *et al.*, 2002). Advancing chemical and flux estimation methods and focus from national and International policies such as the IPCC have provided monitored studies based on more than observations and short term monitoring, and can potentially allow for the widespread investigation of the validity of the Outwelling Hypothesis and impact of wetlands on the surrounding estuaries (Lee, 1995; Bauer *et al.*, 2013).

In the recent decade, important attention has been focused on wetlands, and specifically tidal wetlands through science and policy. In the 2014 supplement to the IPCC 2006, international guidelines were set forth to limit and track the CO_2 emissions from established, created, and rewetted wetlands (Hiraishi et al., 2014). This supplement has stimulated policies around the world for wetland quantification and science, including the United States' National Greenhouse Gas Inventory, Paris Climate

Agreement, and other more regional and local programs around the world (USNGGI; EPA 2017; Hiraishi *et al.*, 2014). For these agreements, large scale and spatially accurate conglomerations of carbon burial, CO_2, CH_4, and N_2O emissions. Currently, there has not been a national quantification of tidal fluxes or carbon loss in the United States.

The purpose of this study was to create order of magnitude carbon flux estimates for the tidal wetlands in the conterminous United States. There are three main objectives for this study:

1. Determine key estuarine regions that transfer high or low amounts of soil organic carbon to other portions of the ecosystems
2. Quantify the amount of soil organic carbon that is being transferred to other portions of the environment, including atmosphere, water, and soil
3. Quantify the relative amount of carbon being transferred at various depths in the soil column to compensate for uncertainties in soil column processes

The objectives for this study will provide insights for coastal and wetland management for restoration and conservation. Key areas determined in this study could lead to more investigation on the influences of tidal wetlands on the nearby estuary and potential management actions that could influence the POC or DOC from the wetland to the estuary.

IV. 2 Methods

In this study, I attempt to quantify and determine soil organic carbon fluxes in the tidal wetlands of the United States. This was done in two different sets of calculations: (i) total carbon loss for the estuarine tidal wetlands, and (ii) the total carbon loss divided among the respective loss processes.

All calculations are computed on multiple depth increments in the soil column due to the uncertainties about relative depths of soil column processes such as microbial influence and soil compaction. Details and suggestions for these uncertainties will be discussed later. The depth increments considered were 0-20 cm, 0-100 cm, and 0-300 cm. The 0-20 cm depth interval includes the majority of root and autotrophic productivity, and is therefore an important target for SOC management and restoration objectives. Ecologically, this is the depth most influenced by soil respiration and atmospheric influences. Microbial activity also occurs deeper in the soil profile, albeit at slower rates compared to the upper organic and A horizons. The other two depths considered, 0-100 cm and 0-300 cm, are calculated to account for microbial influences in deeper portions of the profile. All calculations were determined on an estuarine basin scale due to the restrictions in the availability for certain soil column data, including soil respiration. Estuarine boundaries were acquired from the National Oceanic and Atmospheric Coastal Assessment Framework (CAF). The two major goals of this study were to quantify the total carbon loss, and the relative percentages and amounts of the total carbon loss that are going to neighboring mediums.

IV.2.1 *T_{SOC}: Total Loss of SOC from the Soil Profile*

I begin with the quantity of SOC within the soil column. The SOC quantity was obtained from the blue carbon national geodatabase CoBluCarb (Hinson et al., 2017). All methods and assumptions for this geodatabase can be found in Hinson et al., 2017. T_{SOC}, or total carbon loss, refers to any organic carbon loss between the surface and a specific depth in the soil column. This includes all carbon loss to the atmosphere through respiration, as well as any carbon leached to the water. I defined this in g/m^2/yr as:

$$T_{SOC} = -\sum_{n=1}^{d}[(SOC_n - SOC_d) * a] * uc \qquad (4)$$

where *n* refers to the surface, *d* refers to the lowest depth considered, *a* is the wetland soil accretion, and *uc* is any unit conversion. The SOC at different depths in the column were determined through CoBluCarb (Hinson et al., 2017). The accretion was determined on a one-to-one ratio with relative sea level rise (NOAA). This was accomplished by using NOAA tidal gauge data in the United States. These values were then interpolated to the coastal United States estuarine basins with the average sea level rise height for the estuary. Each estuary had a unique sea level rise value based on location and the interpolation. This ratio is not accurate for all areas, but is an acceptable proxy on a large-scale analysis. The average sea level/accretion calculation for the entire coastal United States was 0.3 mm/yr which falls within the range of accretion in studies across the United States (Turner et al., 2000; Morris et al., 2016; Chmura et al., 2003). Compaction of the soil column can be combined with more accurate regional accretion levels in the future. In this study, compaction is assumed to be consistent and negligible throughout the soil column.

IV.2.2 *E_{SOC} and L_{SOC}: Efflux Due to Soil Respiration and Transport to the Water Column*

The second objective for this study was to determine how much and by which pathway the carbon was leaving the soil column. The first loss pathway from the soil column is to the atmosphere through microbial respiration. In this study, only heterotrophic respiration is considered. The soil respiration or evasion SOC, E_{SOC}, was obtained from values reported in the peer-reviewed literature. Widespread eddy flux tower data is not currently available, and no database or large compilation of soil respiration values in tidal wetlands are known. Most of the soil respiration literature is based on studies conducted in terrestrial environments (natural and agricultural ecosystems), and conversions from these ecosystem types to tidal wetlands in terms of soil respiration is an area that requires more study. The criteria for the literature review included specification of heterotrophic respiration or microbial soil efflux, location within range of determined tidal wetland areas, and consistency of the methods. Due to the paucity of studies located in tidal wetlands, seasonality and time frame of the study was not considered. Most of the values were from previous compilation reviews by Raich and Schlesinger (1992) and Krauss and Whitbeck (2012). There were also site-specific studies that met the criteria for inclusion in the data set (Weston et al., 2011; Wigand et al., 2009). In total, the values for soil respiration were taken from studies that ranged from 1980 – 2007 and had vast differences in the vegetation. There were samples from forested, tidal freshwater, and marshes in the study sites. All of the studies included were on the East Coast. There were no studies from the West Coast and two studies that were not included in the final analysis from the Gulf Coast. The studies from the Gulf Coast were originally considered, but there were only two estuaries studied and multiple samples from different years in the same estuaries, causing contradicting locations in the analysis. These Gulf Coast areas were removed because of the high amounts of error.

Many factors were analyzed with the evasion SOC to create the best relationship possible including temperature, precipitation, coastal location, plant type, salinity, estuarine typology, and wetland type. The best relationship was between evasion SOC and temperature. Since there was no seasonality or time frame considered for the soil respiration literature review, the temperature data was a 30 yr average sourced from Oregon State University PRISM data (Oregon State University PRISM, 2017). The estuarine basins with both soil respiration and temperature values were regressed and the relationship was then calculated to determine the evasion SOC for all the estuarine basins in the United States, as follows:

$$E_{SOC} = 45.23 - (6.1376 \ * temp) \qquad (5)$$

The second direction available for carbon loss is laterally into the nearby water and estuarine basin. Assuming that the total carbon loss (Eq. 1) includes both and only the evasion SOC and lateral flux SOC, I calculated the lateral flux SOC by:

$$L_{SOC} = T_{SOC} - E_{SOC} \qquad (6)$$

These calculations represent the most current information available for these fluxes. As more detailed regional information is available, these calculations can be modified to improve accuracy across the United States. It should be noted that the calculations in this study are analyzed with large assumptions and are not meant to be exact representations. These are order-of-magnitude calculations. The ratios between the areas are important for analyzing key tidal wetland areas around the United States and should be considered accurate for comparison.

IV.3 Results

The fluxes were calculated on three different depths ranges to compensate for the uncertainties for the merits of the different ranges. The ranges include: 0-20cm, 0-100 cm, and 0-300 cm. In table 6, all three ranges are shown for the T_{SOC} and the L_{SOC}. These different depths can considerably change the interpretation of the relationship between the tidal wetlands and the estuarine waters. While in this study, soil respiration is considered at every depth within the soil column, as well as lateral wetland to water fluxes. In the majority of soils, the 0-20 cm depth is considered the productive or the root depth zone (O and potentially A horizons), though depending on the soil, it can be as little as 0-10 cm or down to 0-35 cm depth. In these depths and horizons, soil respiration is known to have a significant influence. In this study, this depth is the most conservative estimate, given the true depths of the soil respiration influence may not end at the depth and can be potentially be at greater depths.

The national average for SOC density and accretion were 0.05 g/cm^3 and 3.05 cm/yr respectively (Table 6). The SOC density values range from 0.002 to 0.098 g/cm^3. The accretion ranges from -0.2 to 8.49 mm/yr.

Table 6. Order-of-magnitude estimates of wetland carbon flux by estuarine drainage area (EDA). T_{SOC} flux is total carbon loss from wetlands within each estuary, E_{SOC} is the portion lost to the atmosphere via soil respiration flux, L_{SOC} is the portion lost to the adjacent estuary. Positive values indicate net carbon influx going into the wetland from the water column. Depth is in centimeters.

Estuary	Wetland Area (km^2)	Density (g/cm^3)	*a* (mm/yr)	T_{SOC} ($g/m^2/yr$) to depth			E_{SOC} ($g/m^2/yr$)	L_{SOC} ($g/m^2/yr$) to depth		
				0-20	0-100	0-300		0-20	0-100	0-300
Albemarle Sound	357.0	0.1	4.0	-0.7	-100.7	-390.2	-53.0	52.3	-47.7	-337.1
Alsea River	3.3	0.0	1.7	25.8	-53.9	-89.1	-22.3	48.1	-31.6	-66.9
Altamaha River	180.6	0.0	2.5	-98.9	-104.1	-196.2	-72.6	-26.3	-31.5	-123.7
Apalachee Bay	208.0	0.0	2.3	-46.2	-104.7	-171.5	-75.8	29.6	-28.9	-95.7
Apalachicola Bay	135.0	0.1	2.4	-13.4	-37.7	-156.0	-76.2	62.8	38.5	-79.7
Aransas Bay	102.9	0.0	5.3	-9.6	-12.8	-54.3	-88.6	79.0	75.8	34.3
Atchafalaya/Vermilion Bays	2465.5	0.1	8.9	-26.7	-174.0	-544.5	-76.7	50.0	-97.3	-467.8
Barataria Bay	1151.0	0.1	8.8	-4.6	-229.0	-524.9	-81.1	76.5	-148.0	-443.8
Barnegat Bay	55.5	0.1	3.6	-33.1	-53.7	-271.4	-29.0	-4.0	-24.6	-242.4

Table 6 Continued

Estuary	Wetland Area (km^2)	Density (g/cm^3)	*a* (mm/yr)	T_{SOC} ($g/m^2/yr$) to depth			E_{SOC} ($g/m^2/yr$)	L_{SOC} ($g/m^2/yr$) to depth		
				0-20	0-100	0-300		0-20	0-100	0-300
Biscayne Bay	131.8	0.1	3.3	-31.0	-101.6	-279.7	-101.7	70.7	0.1	-177.9
Blue Hill Bay	4.5	0.0	2.1	-28.8	-87.5	-143.9	4.8	-33.6	-92.3	-148.8
Bogue Sound	95.1	0.1	3.2	-64.9	-161.5	-271.1	-59.8	-5.2	-101.7	-211.3
Brazos River	116.6	0.0	7.0	-29.7	-258.1	-315.6	-81.4	51.7	-176.6	-234.2
Breton/Chandeleur Sound	1247.1	0.0	7.5	-66.5	-266.2	-450.0	-81.1	14.6	-185.1	-368.9
Broad River	308.8	0.1	3.0	-2.1	-84.0	-324.9	-70.4	68.3	-13.6	-254.4
Buzzards Bay	23.8	0.1	2.7	-33.0	-62.3	-251.2	-18.6	-14.4	-43.7	-232.6
Calcasieu Lake	667.4	0.0	7.1	-9.6	-579.6	-659.3	-75.4	65.8	-504.2	-583.9
Cape Cod Bay	52.5	0.1	3.0	-16.2	-58.0	-244.4	-17.8	1.6	-40.2	-226.6
Cape Fear River	44.5	0.1	2.6	-10.1	-93.6	-232.4	-57.2	47.1	-36.4	-175.2
Casco Bay	9.5	0.0	1.9	-11.9	46.4	-71.4	0.8	-12.7	45.6	-72.2
Charleston Harbor	150.7	0.0	3.2	-4.9	-60.9	-154.7	-66.9	62.0	6.0	-87.8

Table 6 Continued

Estuary	Wetland Area (km^2)	Density (g/cm^3)	*a* (mm/yr)	T_{SOC} (g/m^2/yr) to depth			E_{SOC} (g/m^2/yr)	L_{SOC} (g/m^2/yr) to depth		
				0-20	0-100	0-300		0-20	0-100	0-300
Charlotte Harbor	222.5	0.1	2.8	-26.6	-159.9	-206.4	-95.1	68.5	-64.7	-111.3
Chesapeake Bay	1651.0	0.1	3.2	-39.8	-93.0	-249.4	-39.3	-0.5	-53.7	-210.1
Chincoteague Bay	118.6	0.0	4.0	-25.3	-130.6	-255.4	-41.4	16.2	-89.1	-214.0
Choctawhatchee Bay	38.7	0.1	3.0	-16.9	-30.7	-172.1	-72.7	55.8	42.0	-99.4
Columbia River	104.9	0.0	0.7	-4.2	-10.4	-22.2	-14.7	10.5	4.3	-7.5
Coos Bay	7.5	0.1	1.1	0.4	-13.5	-63.8	-23.3	23.7	9.8	-40.5
Coquille River	1.4	0.0	1.3	-15.1	-19.2	-69.6	-25.0	9.9	5.8	-44.6
Corpus Christi Bay	35.5	0.0	4.8	-12.5	-14.2	-42.4	-90.2	77.6	76.0	47.8
Damariscotta River	1.0	0.1	2.0	-40.7	-57.7	-144.0	0.1	-40.8	-57.8	-144.1
Delaware Bay	699.0	0.1	3.7	-36.3	-12.4	-289.9	-32.8	-3.5	20.4	-257.1
Delaware Inland Bays	26.2	0.0	3.7	-8.8	-93.4	-242.1	-39.1	30.4	-54.2	-203.0
Drakes Estero	2.1	0.0	1.7	-0.4	-24.8	-59.0	-32.8	32.4	8.0	-26.2

Table 6 Continued

Estuary	Wetland Area (km^2)	Density (g/cm^3)	*a* (mm/yr)	T_{SOC} ($g/m^2/yr$) to depth			E_{SOC} ($g/m^2/yr$)	L_{SOC} ($g/m^2/yr$) to depth		
				0-20	0-100	0-300		0-20	0-100	0-300
East Mississippi Sound	170.2	0.1	3.7	43.3	-31.3	-231.6	-72.2	115.5	41.0	-159.3
Eel River	1.6	0.0	2.7	-76.1	-109.1	-123.0	-32.1	-43.9	-77.0	-90.9
Englishman/Machias Bay	7.3	0.1	2.1	-28.1	-57.6	-155.4	7.7	-35.8	-65.3	-163.1
Florida Bay	330.3	0.1	4.1	-110.6	-194.4	-389.7	-105.7	-4.9	-88.7	-284.0
Galveston Bay	347.0	0.0	7.1	-56.3	-246.9	-299.9	-82.1	25.9	-164.8	-217.7
Gardiners Bay	16.5	0.0	2.6	-91.1	-160.2	-174.4	-22.4	-68.7	-137.8	-152.0
Grays Harbor	43.4	0.0	-0.1	1.8	5.2	7.6	-16.1	17.8	21.3	23.7
Great Bay	10.2	0.1	2.3	-15.9	-91.7	-175.4	-5.6	-10.3	-86.1	-169.8
Great South Bay	69.8	0.1	2.8	-12.5	-33.7	-242.2	-25.6	13.0	-8.1	-216.7
Hampton Harbor	15.0	0.1	2.3	-4.7	-21.8	-221.3	-8.3	3.6	-13.5	-213.0
Hudson River/Raritan Bay	90.2	0.1	2.8	-19.5	-63.9	-243.5	-15.9	-3.6	-48.0	-227.7

Table 6 Continued

Estuary	Wetland Area (km^2)	Density (g/cm^3)	*a* (mm/yr)	T_{SOC} (g/m^2/yr) to depth			E_{SOC} (g/m^2/yr)	L_{SOC} (g/m^2/yr) to depth		
				0-20	0-100	0-300		0-20	0-100	0-300
Humboldt Bay	4.7	0.0	3.9	-134.5	-190.5	-212.1	-28.0	-106.5	-162.4	-184.0
Indian River	143.6	0.0	2.9	-64.8	-94.3	-153.2	-92.6	27.8	-1.6	-60.6
Kennebec/Androscoggin River	31.8	0.1	1.5	-6.0	-29.6	-101.5	8.6	-14.7	-38.2	-110.1
Klamath River	0.4	0.0	2.1	-82.9	-100.3	-115.8	-26.6	-56.3	-73.6	-89.1
Long Island Sound	72.6	0.1	2.2	-13.5	-101.7	-217.3	-15.1	1.7	-86.6	-202.2
Lower Laguna Madre	60.0	0.0	3.5	-1.8	-6.7	-18.2	-97.0	95.2	90.3	78.8
Maryland Inland Bays	21.2	0.1	3.9	-117.0	-145.4	-344.1	-39.8	-77.2	-105.6	-304.3
Massachusetts Bay	27.7	0.1	2.5	-21.5	-53.6	-236.5	-15.4	-6.1	-38.2	-221.1
Matagorda Bay	255.0	0.0	6.3	-11.7	-169.4	-214.7	-83.7	72.0	-85.7	-131.0
Mermentau River	776.6	0.0	7.8	-57.1	-528.2	-639.5	-77.2	20.1	-451.0	-562.3

Table 6 Continued

Estuary	Wetland Area (km^2)	Density (g/cm^3)	*a* (mm/yr)	T_{SOC} ($g/m^2/yr$) to depth			E_{SOC} ($g/m^2/yr$)	L_{SOC} ($g/m^2/yr$) to depth		
				0-20	0-100	0-300		0-20	0-100	0-300
Merrimack River	11.0	0.1	2.4	-30.8	-62.7	-275.6	-4.2	-26.5	-58.4	-271.4
Mission Bay	0.2	0.0	2.1	0.0	-15.1	-15.2	-60.5	60.5	45.4	45.3
Mississippi River	423.1	0.0	8.1	-114.6	-216.8	-428.7	-82.8	-31.8	-134.0	-345.9
Mobile Bay	72.6	0.0	3.1	-22.2	-25.2	-145.1	-73.4	51.2	48.2	-71.7
Monterey Bay	6.7	0.0	1.3	-0.5	-1.4	-15.1	-46.2	45.7	44.8	31.1
Morro Bay	2.0	0.0	1.0	-0.1	-14.7	-16.5	-43.6	43.5	29.0	27.1
Muscongus Bay	3.1	0.1	2.0	-50.3	-63.6	-158.0	0.8	-51.1	-64.3	-158.7
Narragansett Bay	15.0	0.1	2.6	-18.4	-119.8	-232.1	-17.3	-1.1	-102.6	-214.8
Narraguagus Bay	10.4	0.1	2.1	-28.8	-49.2	-165.9	6.4	-35.1	-55.5	-172.3
Nehalem River	6.1	0.0	0.4	-1.7	-4.8	-22.5	-14.3	12.6	9.5	-8.2
Netarts Bay	0.9	0.1	1.7	-4.4	-14.4	-99.5	-19.1	14.7	4.8	-80.4
New Jersey Inland Bays	431.0	0.1	3.9	-17.5	-10.1	-386.3	-31.7	14.2	21.5	-354.6

Table 6 Continued

Estuary	Wetland Area (km^2)	Density (g/cm^3)	*a* (mm/yr)	T_{SOC} (g/m^2/yr) to depth			E_{SOC} (g/m^2/yr)	L_{SOC} (g/m^2/yr) to depth		
				0-20	0-100	0-300		0-20	0-100	0-300
New River	27.7	0.0	2.7	-6.2	-88.6	-204.3	-57.8	51.6	-30.8	-146.5
Newport Bay	1.4	0.1	1.6	-3.1	-6.4	-6.8	-65.9	62.8	59.6	59.2
North Ten Thousand Islands	924.9	0.0	3.2	-19.1	-33.9	-266.6	-101.3	82.2	67.4	-165.3
North/South Santee Rivers	163.5	0.1	3.1	-18.7	-30.3	-166.6	-65.1	46.3	34.7	-101.6
Ossabaw Sound	222.4	0.1	2.7	-4.1	-61.3	-259.3	-70.7	66.5	9.4	-188.6
Pamlico Sound	648.5	0.1	3.5	-19.5	-117.2	-315.9	-58.8	39.3	-58.4	-257.1
Passamaquoddy Bay/Cobscook Bay	3.3	0.0	2.0	-26.5	-61.5	-124.7	9.6	-36.1	-71.0	-134.3
Penobscot Bay	6.2	0.1	1.9	-30.7	-46.9	-146.0	6.1	-36.8	-53.0	-152.1
Pensacola Bay	61.3	0.1	2.6	-5.6	-95.4	-257.5	-72.6	67.0	-22.7	-184.8

Table 6 Continued

Estuary	Wetland Area (km^2)	Density (g/cm^3)	*a* (mm/yr)	T_{SOC} ($g/m^2/yr$) to depth			E_{SOC} ($g/m^2/yr$)	L_{SOC} ($g/m^2/yr$) to depth		
				0-20	0-100	0-300		0-20	0-100	0-300
Perdido Bay	14.5	0.0	2.9	-2.3	-38.8	-137.5	-73.8	71.5	35.0	-63.7
Plum Island Sound	40.5	0.1	2.4	-5.9	-54.9	-239.0	-12.3	6.4	-42.7	-226.7
Puget Sound	46.0	0.0	1.1	-24.7	-32.0	-53.6	-11.8	-12.8	-20.1	-41.8
Rio Grande	0.4	0.0	4.4	-0.5	-13.9	-25.2	-99.6	99.1	85.7	74.4
Rogue River	0.6	0.0	0.8	-1.9	-24.2	-30.5	-25.9	24.0	1.8	-4.6
Rookery Bay	75.1	0.1	2.8	-13.9	-21.5	-244.1	-99.1	85.2	77.6	-145.1
Sabine Lake	593.2	0.0	6.8	-83.6	-483.1	-588.1	-74.5	-9.1	-408.6	-513.7
Saco Bay	17.9	0.1	1.8	-19.2	43.1	-95.2	6.4	-25.6	36.7	-101.6
San Antonio Bay	101.3	0.0	5.6	-13.2	-28.8	-58.1	-86.2	73.0	57.4	28.1
San Diego Bay	1.5	0.0	1.9	0.0	-32.9	-43.4	-57.6	57.6	24.7	14.2
San Francisco Bay	212.0	0.1	1.3	-7.8	-29.9	-102.1	-51.8	43.9	21.9	-50.3
San Pedro Bay	2.7	0.0	1.4	0.0	-0.1	-2.4	-61.2	61.2	61.2	58.9

Table 6 Continued

Estuary	Wetland Area (km^2)	Density (g/cm^3)	*a* (mm/yr)	T_{SOC} (g/m^2/yr) to depth			E_{SOC} (g/m^2/yr)	L_{SOC} (g/m^2/yr) to depth		
				0-20	0-100	0-300		0-20	0-100	0-300
Santa Monica Bay	0.6	0.0	1.5	-2.9	-2.9	-71.3	-61.6	58.7	58.7	-9.7
Sarasota Bay	10.2	0.0	2.8	-101.5	-158.3	-196.2	-94.7	-6.8	-63.6	-101.5
Savannah River	271.6	0.1	3.0	-38.7	-74.9	-237.4	-71.0	32.3	-3.9	-166.4
Sheepscot Bay	9.0	0.1	1.9	-31.1	-37.7	-168.3	1.0	-32.1	-38.7	-169.3
Siletz Bay	2.1	0.0	2.1	-13.5	-18.0	-94.1	-18.5	4.9	0.4	-75.7
Siuslaw River	5.9	0.1	1.3	-5.1	-30.6	-71.8	-23.6	18.4	-7.0	-48.2
South Ten Thousand Islands	893.8	0	3.77	NA	NA	NA	-104.03	NA	NA	NA
St.Andrew Bay	42.5	0.1	2.9	-31.0	-72.3	-105.7	-76.1	45.1	3.8	-29.6
St.Andrew/St.Simons Sounds	551.8	0.0	2.4	-109.9	-114.0	-297.0	-73.3	-36.6	-40.7	-223.7

Table 6 Continued

Estuary	Wetland Area (km^2)	Density (g/cm^3)	*a* (mm/yr)	T_{SOC} ($g/m^2/yr$) to depth			E_{SOC} ($g/m^2/yr$)	L_{SOC} ($g/m^2/yr$) to depth		
				0-20	0-100	0-300		0-20	0-100	0-300
St.Catherines/Sapelo Sounds	532.3	0.1	2.6	-1.7	-52.6	-280.7	-73.6	71.9	20.9	-207.2
St.Helena Sound	379.9	0.1	3.0	-6.0	-76.9	-300.2	-67.9	61.9	-9.1	-232.4
St.Johns River	78.7	0.1	2.4	-1.9	-189.8	-230.6	-83.3	81.4	-106.5	-147.4
St.Marys River/Cumberland Sound	184.7	0.1	2.3	-44.0	-48.4	-256.8	-78.0	34.0	29.5	-178.8
Stono/North Edisto Rivers	177.8	0.0	3.1	-1.6	-4.4	-11.4	-69.7	68.1	65.3	58.2
Suwannee River	112.4	0.0	2.1	-32.2	-93.8	-150.4	-79.5	47.3	-14.3	-71.0
Tampa Bay	88.3	0.0	2.7	-42.7	-61.1	-111.3	-93.9	51.2	32.8	-17.4
Terrebonne/Timbalier Bays	1155.6	0.0	8.9	-30.0	-159.9	-444.8	-81.4	51.4	-78.4	-363.3

Table 6 Continued

Estuary	Wetland Area (km^2)	Density (g/cm^3)	*a* (mm/yr)	T_{SOC} (g/m^2/yr) to depth			E_{SOC} (g/m^2/yr)	L_{SOC} (g/m^2/yr) to depth		
				0-20	0-100	0-300		0-20	0-100	0-300
Tijuana Estuary	1.8	0.0	1.5	0.0	-14.8	-21.1	-48.0	48.0	33.2	26.9
Tillamook Bay	5.0	0.1	1.1	0.2	-5.5	-61.5	-15.2	15.4	9.8	-46.2
Tomales Bay	4.1	0.0	1.7	-13.8	-27.5	-40.8	-41.2	27.4	13.7	0.4
Umpqua River	7.2	0.0	1.2	-0.8	-28.1	-58.0	-24.7	23.8	-3.4	-33.3
Upper Laguna Madre	9.7	0.0	4.0	-4.5	-12.6	-26.0	-92.3	87.9	79.7	66.3
Waquoit Bay	1.4	0.1	2.7	-11.4	-25.6	-189.5	-18.0	6.6	-7.7	-171.5
Wells Bay	3.5	0.0	2.8	-9.5	-13.7	-123.0	-4.0	-5.5	-9.7	-119.0
West Mississippi Sound	780.3	0.1	7.2	55.9	-94.5	-476.5	-74.9	130.8	-19.7	-401.6
Willapa Bay	44.5	0.0	-0.2	1.4	7.2	10.2	-16.7	18.1	24.0	27.0
Winyah Bay	339.9	0.1	2.9	-10.9	-32.9	-190.5	-60.1	49.2	27.2	-130.4
Yaquina Bay	2.1	0.0	2.1	-10.3	-19.8	-96.6	-21.9	11.6	2.1	-74.7
National Average	197.9	0.0	3.0	-23.8	-79.3	-190.7	-48.9	24.6	-30.9	-142.3

Overall, the national average ranges from -23.83 (0-20 cm depth) to -190.74 (0-300 cm depth) grams of soil organic carbon loss across the all the soils in tidal wetlands across the United States. Out of the 115 total estuaries, there are 7, 4, 2 and estuaries that are net importers of OC from the surrounding waters at 0-20 cm, 0-100 cm, and 0-300 cm depth respectively. All negative values are considered losses from the wetland into surrounding mediums such as the atmosphere and the water. The national average range widely depending on what depth is considered. The total flux of OC from wetlands to the waters of the estuaries of the continental US was calculated to lie between 1.00 Tg yr^{-1} and -5.79 Tg yr^{-1}. This range is based on the variability between the estimates of L_{SOC}, using the upper 20 cm of depth versus the full 300 cm of depth. The central value at 100 cm was -1.26 Tg yr^{-1}.

The E_{SOC} ranges from -105.72 to 9.56 $g/m^2/yr$. The national average is -48.94 $g/m^2/yr$. Only 11 estuaries included positive soil respiration values, implying that these 11 estuaries were taking in more carbon than they released back to the atmosphere. As seen in Fig.1, the E_{SOC} comes from the surface but can potentially come from lower in the soil column to the atmosphere. This flux is typically only measured in short depth studies, in ranges from the surface to approximately 35 cm depth. This can be seen in the corresponding graph for process (1) (Fig.10). There is a large amount of E_{SOC} at the surface of the soil column, but after the mid-A horizon (or the top percent of the soil column) the amount of respiration is stagnant. While there could be potential respiration influences in the lower horizons, there have been very little to no studies on the impact of lower horizon respiration from potential anaerobic microbe communities. Soil respiration influences were extended down the soil column, though it is not known if this is a valid process to that depth. If this assumption is not valid, the T_{SOC} for the lower horizons will be completely due to the L_{SOC} into the water.

The L_{SOC} ranges from -504.22 to 90.31 g/m^2/yr. This flux had the widest range of any that were calculated in this study. There are 80 estuaries that at 0-20 cm depth, are net importers of OC from the water. By 0-300 cm depth however, there are only 15 estuaries. There is still a lot of uncertainty in the factors considering lateral flux with depth, as show in the drastic changing in importing and exporting in total estuaries. In Figure 1, the black arrows on the side of the soil column show the amount of flux with depth in the column. As the depth increases, there is more compaction and less carbon, creating smaller and widespread arrows that transfer carbon to the water since the carbon will be leaching from the soil slower due to the closeness of the soil aggregates and the heaviness of the column above. In the graph for the L_{SOC} (process (3) in Fig. 10), the curve is similar to E_{SOC}. Both of these processes decrease exponentially with the depth, though while E_{SOC} becomes constant due to the potential lack of microbial communities, the L_{SOC} does not stop, but just continues at a slower and still decreasing rate until the bottom of the column.

There are multiple different factors that contribute to the carbon loss to the water in each soil type, including bulk density, porosity, amount of compaction, recalcitrant aggregates including carbon, tidal frequency and strength, and the amount of other disturbance such as humans, wildlife or storms. In this study, compaction is assumed to be a 1:1 ratio with accretion. Currently, there is no equation that compensates for compaction on a large or small scale. If compaction is included in the study at all, it is simply to only consider cores with relatively the same compaction, so the influence does not have to be considered.

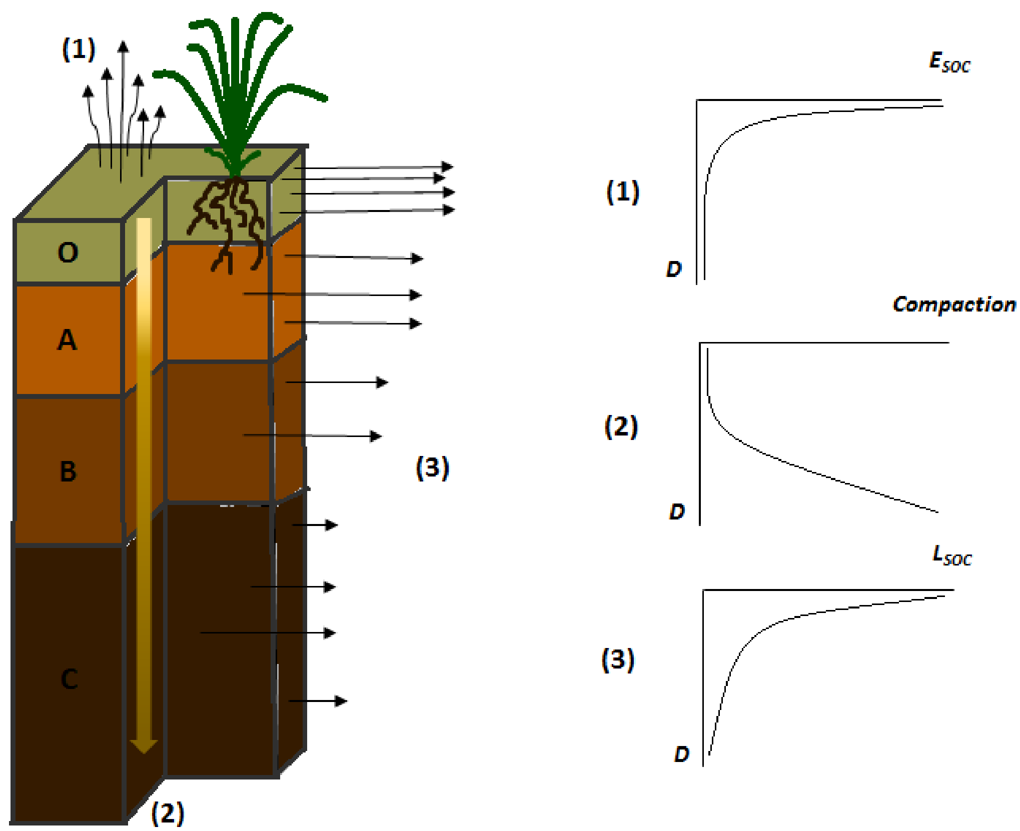

Figure 10. *A visual representation of vital physical and chemical processes concerning organic carbon loss in the soil column by horizon. D represents the depth. Location (1) indicates* E_{SOC} *or soil respiration from the O to A horizons. (2) indicates compaction from the surface to the bottom of the column. Darker colors indicate greater compaction within the horizons. (3) indicates* L_{SOC} *or lateral flux to the surrounding water. Due to compaction, particle size and amount differs in lower horizons.*

In Figure 10, soil compaction is show as a gold arrow. As the depth increases, so does the compaction and the darkness of the arrow. The arrow is pointed down in the direction of the compaction as well. As seen in the corresponding graph, the compaction is stagnant and has little influence at the surface of the soil column, but as the depth increases, the rate of compaction also increases. There is more organic matter at the surface of the column and decreases as the depth increases, causing compaction to be more influential in the lower horizons. This is one of the main causes of compaction and can be seen in the colors of the horizons as well.

Key areas of interest are more noticeably observed in Figure 11. The order of this figure from top to bottom is E_{SOC}, L_{SOC}, and T_{SOC} and represents all the estimations at 100 cm depth. The estuaries that had the most positive L_{SOC} (import of OC from the water to the wetland) at the 100 cm depth, included the Tijuana Estuary and San Diego Bay in Southern California, and the Rio Grande and the Lower Laguna Madre in Texas but strongly negative E_{SOC} (Fig. 11a). The estuaries that had the most negative L_{SOC} values at the 0-100 cm depth (export of OC from the wetland to the water) can be found in the Chenier Plain of Louisiana and Texas, and in Southern Louisiana. Ranked in order from the most negative L_{SOC} at the 100 cm of depth: Calcasieu Lake, Mermentau River, Sabine Lake, Breton/Chandeleur Sound, Brazos River, and Galveston Bay. In areas such as Atchafalaya/Vermillion Bays, Barataria Bay, West Mississippi Sound, and Terrebonne/Timbalier Bays, these estuaries have strong negative T_{SOC} values or high OC loss from the soil (Fig. 11c). These estuaries are in warm locations in the Southern United States, stimulating high soil respiration rates and strong negative E_{SOC} levels (Fig. 11a). In this case, the mass balance of Eq. 4 results in a great amount of lateral export into the water column (Fig. 11b).

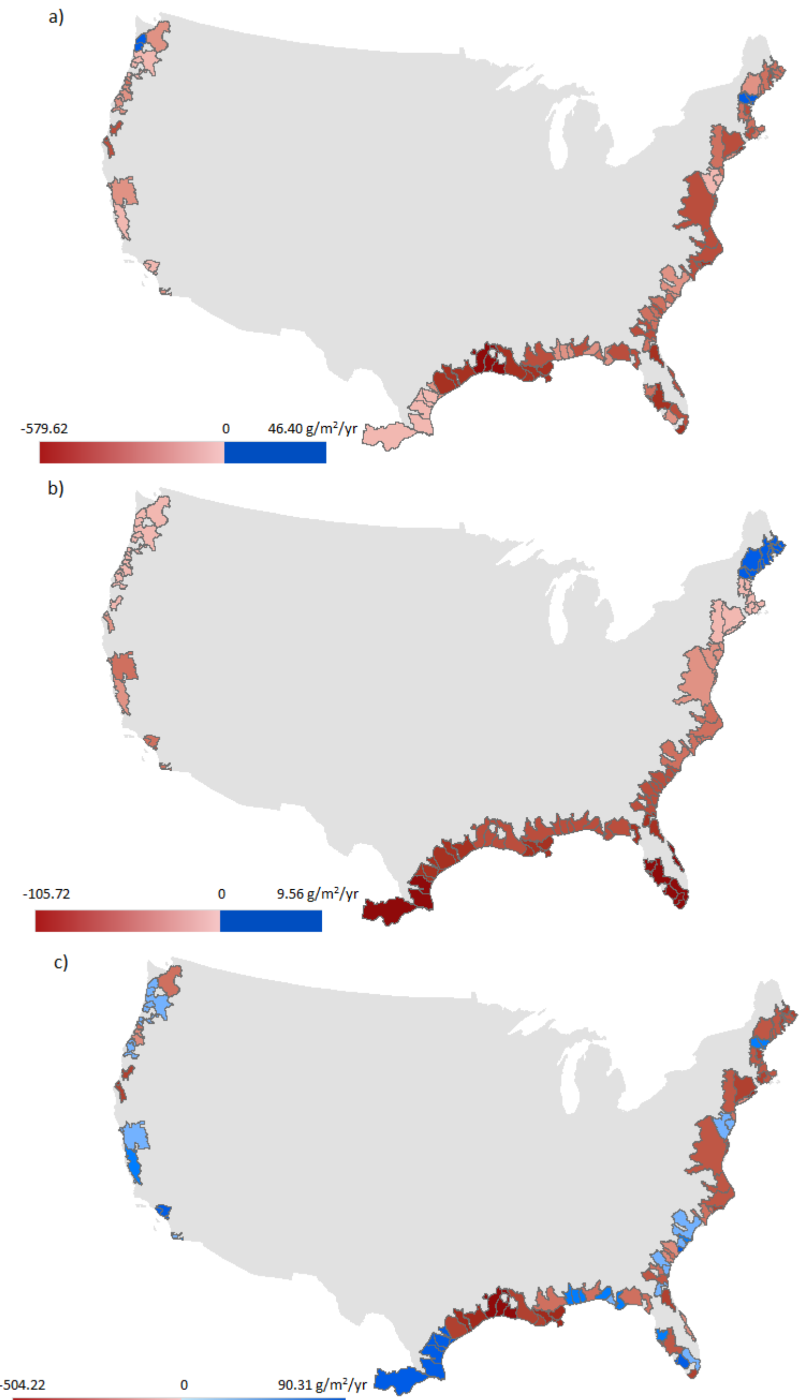

Figure 11. *Complete estuarine maps showing the fluxes at 0-100 cm depth a) T_{SOC} b) E_{SOC} and c) L_{SOC}*

IV.4 Discussion

The E_{SOC} and L_{SOC} are additive and create the T_{SOC} map on the bottom of Figure 11 as can be seen in Eq. 4 as well. For example, Tijuana Estuary and San Diego Bay in Southern California, and the Rio Grande and the Lower Laguna Madre have little organic matter within their typically sandy soils and warm temperatures with sparse yet extreme precipitation occurrences, leading to minimal T_{SOC}. Warm temperatures are related to high respiration rates and in this study, negative E_{SOC} (Zogg *et al.,* 1995; Jenkinson *et al.,* 1991; Davidson and Janssens, 2006; Valiela, 1985). With little T_{SOC} and a large negative E_{SOC}, the mass balance will indicate a large positive L_{SOC}, creating the image that these dry climate wetlands are potentially net importers of OC from the surrounding environments, instead of producing their own.

The evasion SOC was created in this study using a relationship between temperature and heterotrophic respiration. The degree in which temperature is the limiting factor for organic matter decomposition in wetlands, or CO_2 efflux, is still being debated (Davidson and Janssens, 2006). Our study was limited based on peer reviewed literature tidal wetland soil respiration values. Due to these restrictions and the low amount of published soil respiration values in tidal wetlands, the relationship was relatively weak, but this work could be expanded and improved. A source of inconsistency that provided uncertainty throughout the calculation of the E_{SOC} is the accurate depth of influence. The depths of soil respiration studies are variable, with no validation on the depths or how deep soil respiration influences the chemical processes within the soils. Many factors were included in this analysis as well, but as stated in the methods, temperature provided the greatest and most reliable relationship. According to a study by Chambers et al. (2013), introduction of sulfate (SO_4^{2-}) through salinity and frequency of inundation pulses were the causes for changes in CO_2 flux and carbon lost in freshwater intertidal and salt marshes respectively in Florida. Salinity and saltwater intrusion, especially in tidal freshwater wetlands, is very important in the rate of organic matter decomposition and bacterial abundance and can also be significant as coastal areas alter with increasing climate change (Morrissey *et al.*, 2014). Amount of inundation and anaerobic microbial activity can potentially have a sizable impact (Kirwan and Megonigal, 2013). In future calculations of carbon loss, wetland type, based on salinity and litter type (vegetation) should be included and can provide important variations among regions, globally and nationally. Eutrophication and precipitation are also known to affect the soil respiration in wetlands, but to the extent these produce negative feedbacks in regularity needs further investigation (Valiela, 1985; Raich and Potter, 1995).

Compaction of the soil over time and at depth likely produces variable effects on the density of SOC within discrete bins of soil within the soil column. The compaction process also alters the accumulation rate, but I ignored this process by simply using a single accretion value. Compaction could explain uncertainties within the soil column that created unexpected results as the L_{SOC} was calculated at multiple depths. The West Mississippi Sound had both the most strongly positive L_{SOC} values at the 0-20 cm depth (net import) and one of the more strongly negative L_{SOC} values at the 100 cm and 300 depths (net export). The East Mississippi Sound was somewhat similar, though less pronounced. This location indicates an uncertainty that expresses the need for further understanding of soil processes at depths lower than 20 cm. As depth increases, there is greater compaction within the soil column. Compaction is an understudied phenomenon in tidal wetlands that can account for large implications in lateral soil fluxes, involving many physical, biological, and chemical processes that diminish the vertical dimensions of the soil column (Brain *et al.*, 2012) (Fig 10). Compaction is highly associated with sediment accretion, but in many studies of accretion, especially in salt marshes, the effects of compaction are assumed to be negligible, potentially giving an accelerated appearance of sea level rise and accretion (Brain *et al.*, 2012). However, there are some studies that do factor compaction into the soil processes, obtaining specific soil cores that have consistent compaction (Callaway *et al.*, 2012;). Soil compaction and accretion are both highly variable and time dependent processes, especially in tidal influenced wetlands, and are susceptible to severe impacts from sea level rise and climate change (Tuner *et al.*, 2006; Kirwan and Megonigal, 2013). On large spatial scales, compensating for compaction and accretion interactions can be extremely difficult. Changes in decomposition, water influxes (flooding), compression, and organic matter can all affect the bulk density downcore and may not be able to be compensated for on large scales, causing over- or under- estimations of accretion and compaction (Turner *et al.*, 2006). In the many of the soils of the United States Atlantic and Gulf of Mexico coasts, the soil column is composed of organic rich sediments and much of the space is taken up pore space and water, causing a variable sediment accretion rate based on organic matter accumulation (Turner *et al.*, 2006; Kirwan and Megonigal, 2013). Organic soil inputs lack the relatively rigid structure of the large inorganic inputs in mineral soils, creating a volumetrically variable vertical soil column through both accretion and compaction (Turner *et al.,* 2006).

While this study appropriately describes the carbon loss within the soil column, the arenas in which the loss is quantified (laterally and through respiration) does not include riverine processes that may alter these quantities. Riverine influence can provide important SOC inputs to the soil that could offset or mitigate the quantity of SOC that is removed as Eq. 4 is calculated across depth (Bauer *et al.,* 2013). The relatively low T_{SOC} losses of the US West Coast as compared with the East and Gulf Coasts and the relatively high frequency of positive L_{SOC} in estuaries that contain large rivers or inflow sources such as the Columbia River, portions of the Everglades, and the around South Carolina and Georgia could potentially be due to the absence of large deltaic plains composed of wetlands high in organic materials.

There are many complications and further research needed to create a more accurate representation of wetland SOC fluxes. The value of these estimations is not a definitive numerical value, but rather to expose the areas of science that need further study. There are factors considered in this study, but many more than can be refined and included such as compaction (Fig.10), riverine inputs, and biological inputs such as depth of microbial respiration influences and autotrophic respiration. The greatest source of potential error that I found concerns OC variability across ‘depth’ and ‘time’. These include the depth to which soil respiration occurs, the compaction rates of soil across depth, and the integration of soil depth across time increments. Slight difference in these depth considerations can change the resulting fluxes drastically, even changing an estuary from import to export. To improve this work in the future, more biogeochemical research will be required on the depth and time limits over which these linked wetland-estuary processes operate.

In summary, wetlands play an important role in the nutrient cycles in estuaries, whether it be from export or import. There are many unknowns in large scale flux calculations, but both management and estuarine and wetland science can be impacted by accurate quantification. As policies and ecosystems evolve in response to climate change, wetlands will become a vital opportunity for understanding and potentially mitigating negative effects, for both ecosystems and society.

CHAPTER V

CONCLUSIONS

Across the continental United States, there is a vast amount of variation in soil organic carbon in tidal wetlands, even when considered on multiple scales. The soil organic carbon was quantified spatially and vertically with high resolution, creating a large geodatabase, CoBluCarb. The source data for CoBluCarb was both National Wetlands Inventory including all four tidal wetland types (estuarine emergent vegetation, estuarine shrub-scrub, estuarine forested, and tidal freshwater) and SSURGO, containing all 8714 soil types. This geodatabase is important for a multitude of purposes, including helping to shape national, regional, and local scale policies. There are three different scales in which examining correlations of environmental aspects and the soil organic carbon density in tidal wetland soils. In the United States, the regional scale has proven to be the most descriptive, providing important differences in the key aspects as well as categories of aspects that could influence carbon sequestration on the different coasts. Coinciding with carbon storage, carbon loss plays a large role in tidal wetlands and carbon sequestration quantification. There are multiple ways for carbon to be lost from the soil, through respiration and lateral flux mainly, though at what depth and intensity in the column is unknown. Soil organic carbon flux quantifications need to consider many soil processes that are not fully understood. Due to these underlying issues and the lack of widespread site-specific data, the pressing demand for carbon flux measurements are still primitive and quantifiable on order-of-magnitude levels.

Further investigation into the conditions that promote high soil organic carbon densities and more accurate flux measurements can lead to better wetland conservation, more adaptive and quality management decisions, and greater scientific understanding on how the wetland system affects carbon deposition and storage in other ecosystems.

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