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The spatial distribution of soil organic carbon in tidal wetland soils of the continental United States

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Abstract

Tidal wetlands contain large reservoirs of carbon in their soils and can sequester carbon dioxide (CO_2) at a greater rate per unit area than nearly any other 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, state, local, and project-level evaluation of CO_2 sequestration credits, we developed a geodatabase (CoBluCarb) and high-resolution maps of soil organic carbon (SOC) distribution by linking National Wetlands Inventory data with the U.S. Soil Survey Geographic Database. For over 600,000 wetlands, the total carbon stock and organic carbon density was calculated at 5-cm vertical resolution from 0 to 300 cm of depth. Across the continental United States, 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 wetland area, twice as much carbon as the most recent national estimate. Approximately 75% of this carbon was found in estuarine emergent wetlands with freshwater tidal wetlands holding about 19%. The greatest pool of SOC was found within the Atchafalaya/Vermilion Bay complex in Louisiana, containing about 10% of the U.S. total. The average density across all tidal wetlands was 0.071 g cm^{-3} across 0–15 cm, 0.055 g cm^{-3} across 0–100 cm, and 0.040 g cm^{-3} at the 100 cm depth. There is inherent variability between and within individual wetlands; however, we conclude that 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 perturbations. This Tier 2-oriented carbon stock assessment provides a scientific method that can be copied by other nations in support of international requirements.

KEYWORDS

blue carbon, coastal, mangroves, National Wetlands Inventory, salt marsh, soil organic carbon, Soil Survey Geographical, tidal, wetlands

1 | INTRODUCTION

Tidal wetlands are among the most biologically productive and societally valuable ecosystems in the world (Barbier et al., 2011; Costanza et al., 1997; Howard et al., 2017; Martínez et al., 2007), yet they continue to be lost at a global rate of approximately 1.5% annually (Hopkinson, Cai, & Hu, 2012; Pendleton et al., 2012). The pace and scale of these losses has focused global attention on the strategic need for initiatives that promote conservation and sustainable restoration of the physical landscape (Day et al., 2007; Howard et al., 2017). 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, Hoyt, Isensee, Telszewski, & Pidgeon, 2014).

Given the global extent of tidal wetlands and their high levels of productivity, blue carbon is a potential sequestration component for atmospheric carbon dioxide (CO_2) (Chmura, Anisfeld, Cahoon, & Lynch, 2003; Laffoley & Grimsditch, 2009; Mcleod et al., 2011). Sequestration per unit area in these systems is estimated to be as much as 3–50 times greater than that of rainforests (Breithaupt, Smoak, Smith, Sanders, & Hoare, 2012; Bridgham, Megonigal, Keller, Bliss, & Trettin, 2006; Howard et al., 2017; Nellemann et al., 2009). 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, Pendleton, Jenkins, & Sifleet, 2011). Within the United States, tidally influenced 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, Megonigal, Keller, Bliss, & Trettin, 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 tidal 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. Commonly referred to as the U.S. National Greenhouse Gas Inventory (USNGGI; EPA 2017), the EPA is applying new procedures provided by IPCC (Intergovernmental Panel on Climate Change) during the recent Paris Climate Agreement to recognize changes in carbon stocks associated with human activities (Hiraishi et al., 2014). In wetlands, this is accomplished through three Tiers, each describing varying levels of detail, accuracy, and modeling methods (Hiraishi et al., 2014). A Tier 2 analysis, whereby country-specific activity data and emissions factors are applied, is being tested in this study (Hiraishi et al., 2014). In both management and science, 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 United States.

The carbon in tidal wetlands also can have private economic value if managed for sale through offset transactions (Crooks, Emmett-Mattox, & Findsen, 2010; Duarte, Middelburg, & Caraco, 2005; Needelman et al., 2012; Wylie, Sutton-Grier, & Moore, 2016). In total, ecosystem management projects across the United States 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, Alder, Nakamura, Kenchington, & Tamelander, 2013; Lau, 2012; Ullman, Bilbao-Bastida, & Grimsditch, 2012), neither the spatial distribution of carbon nor the monetized credits themselves should be considered homogeneous (Marland, McCarl, & Schneider, 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, Findsen, Igusky, Orr, & Brew, 2009; Nahik & Fennessy, 2016).

Our overall objective was to delineate the geographic distribution of soil organic carbon (SOC) across the tidal wetlands of the continental United States 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; Jenny, 1941). We sought to compare SOC across a variety of categories, such as wetland types, U.S. states, coastlines, and estuarine basins. In order to accomplish this task, we created the CoBluCarb database by combining National Wetlands Inventory (NWI) data with U.S. Department of Agriculture (USDA) Soil Survey Geographical (SSURGO) data.

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, we evaluated its usefulness by comparing it to the literature. Finally, we mapped, summarized, and characterized the carbon distribution across various categorizations of tidal wetlands and through the United States' estuarine basins. To define estuarine extents, we 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, we present only the results from the EDAs and summarize all of the CDAs with a single presented value. In general, the majority of the CDAs individually contain extremely small areas of wetlands and thus are better summarized as a single value.

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, Carter, Golet, & LaRoe, 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, we created the specific requirement that "S", "R","T", or "V" modifiers, temporarily, seasonally, semi-permanently, or permanently flooded tidal regimes respectively, had to be listed in the NWI palustrine wetlands to be considered tidal in CoBluCarb. The wetlands of interest for this study were defined by the interests of the U.S. National Greenhouse Gas Inventory (EPA 2017) to included tidal freshwater wetlands, and thus go beyond those defined as 'blue carbon' in the IPCC Wetlands Supplement (Hiraishi et al., 2014). 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, may be present throughout the tidal wetlands. ArcGIS was used to extract relevant wetland classes from the full NWI database, then any spatial overlap of individual wetland polygons was removed to avoid double-counting through manual visualization, matching entries in their attribute tables when combined with SSURGO, and by categorizations within original Python scripts. We then combined the extracted files into four tidal wetland classes estuarine emergent vegetation (EM; largely equivalent to brackish to saline salt marsh), estuarine shrub-scrub (SS; dominated by shrubs and small mangroves), estuarine forested (FO; largely mangroves), and freshwater tidal (FT; including herbaceous, shrub, and forest vegetation).

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, Borgnis, Turner, & Milan, 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 laboratory 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 complete and fully 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). MUs can be anywhere in the United States, as long as the soil is of the same soil type. 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, Louisiana (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 we 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 we do not present this work here).

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} *100%), though we can easily transform this organic matter percentage into the organic matter fraction (OMF) by dividing by the total 100% (g_{om}/g_{soil}). Together with bulk density, BD (g_{soil}/cm^3) and the van Bemmelen constant, $v = 0.58$ *(g_{soc}/g_{om}), 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

$$\mathrm{OCD}_{cb(h)} = \mathrm{OMF}_{cb(h)} \cdot \mathrm{BD}_{cb(h)} \cdot \nu \tag{1}$$

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 (Keller et al., 2012; Pribyl, 2010; Zhong & Xu, 2011), 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

$$SOC_{cd} = \sum_{b=1}^{d}\sum_{h=1}^{n} OCD_{cb(h)} I_{cb(h)} \quad (2)$$

where $I_{cb(h)}$ 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 $I_{clovely,40(1)} = 1$ while $I_{clovely,40(2)} = I_{clovely,40(3)} = 0$. Similarly $I_{clovely,90(2)} = 1$ while $I_{clovely,90(1)} = I_{clovely,90(3)} = 0$. 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

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

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, we listed all 8,714 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. Our methods for dealing with the various types of missing information are described in detail in the Data S1 section.

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).

2.4 | Evaluation of CoBluCarb

For evaluation and comparison, we regressed the carbon density values from our database against those found at the same spatial location as sourced from Ouyang & 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, we found the correlation between our dataset and these values to be potentially significant though relatively low when considering discrepancies innate to the current field data available (see Data S2, and Fig. S1). The scatter and variance were likely attributable to at least four sources of error: (i) the locations of the literature-derived field samples were coarse and imprecise; (ii) 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; (iii) the literature-derived values were acquired using many different methods, with variable degrees of accuracy and precision; (iv) 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, Edwards, Crooks, & Reyes, 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. We concluded that though it likely contains mean bias (See extensive discussion of error, bias, and variance in Data S2), CoBluCarb likely provides the more spatially accurate, depth-explicit, methodologically consistent, and widely applicable stock estimate for the continental U.S. wetlands.

2.5 | Spatial distribution of tidal wetland carbon

For visual presentation, we populated the NWI records/polygons with the calculated values from CoBluCarb. The final spatial database (as mapped data; Figure 1) 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. 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, we 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. While this estimation is the most accurate given the available data, it does not take into account all NWI wetland area. The high-limit was based on an assumption that these missing cases were similar to the

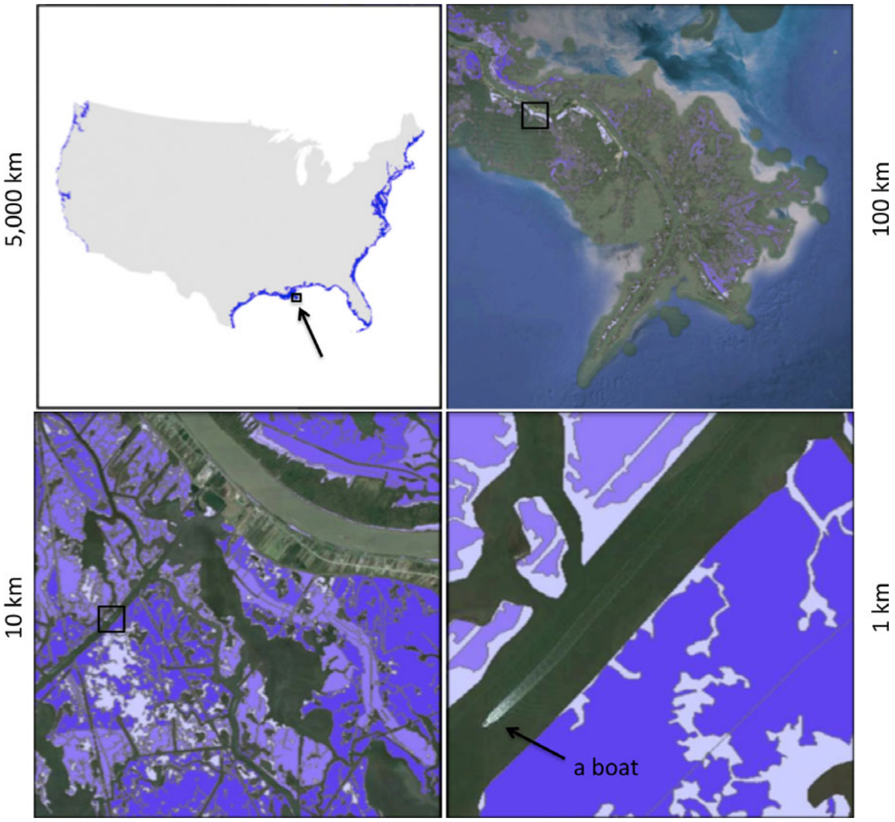

FIGURE 1 Example of database spatial resolution, depicting average carbon density of the upper 15 cm of depth across all saltwater wetland polygons in view. Black boxes denote inset views for subsequent panels. Colors represent varying carbon densities (shown only as example of spatial resolution here) [Colour figure can be viewed at wileyonlinelibrary.com]

known cases within the same geographical extent on an area-weighted basis. Assuming missing data areas are not outliers for SOC stock or density, a high estimation can be calculated by simply scaling the carbon stock from the known area (low estimation) to the carbon stock of the total area which includes the missing data locations (high estimation). This value should be seen as a liberal estimate, though it could either over-estimate or under-estimate based in principle. This estimation range, with the low estimation as a current known carbon and the high estimation as the potential carbon, can give a better snapshot of differences between locations, without hindering from missing data. This estimation is not a confidence interval. This estimation is simply the carbon stock with the current spatial locations and soil data, with an estimation for the amount of carbon if there were no missing data whatsoever. The current carbon stock is an under estimation due to the fact that all wetland areas are not included. The high estimation could be an over estimation or still an under estimation as all areas are included, but this high estimation does make an assumption of stationarity about the soil properties in these missing areas.

For the area-weighted average carbon density across each of the two depths, as well as at the 100 cm depth itself, we summarized only the individual polygons/records with complete information.

3 | RESULTS

Across the tidal wetland soils area of the continental United States, the data are 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 (Figure 2).

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 estuarine emergent vegetation wetlands is somewhat normally distributed by areal coverage and notably covers much larger areas than the other vegetation types (Figure 3).

The distribution across the Gulf Coast is also somewhat normal, while the East Coast is more bi-modal (Figure 4). The West Coast has relatively little area of tidal wetland soils.

Most SOC is stored in estuarine emergent wetlands (Table 1), 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. Soil in estuarine 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. The average organic carbon density decreased with greater soil depth for all wetland types (Figure 5a). However, only the estuarine emergent type displayed a noticeably decreasing standard deviation with greater depth (Figure 5b), with its inflection point at around 25–30 cm of depth (roughly expected as the rooting depth for herbaceous cover).

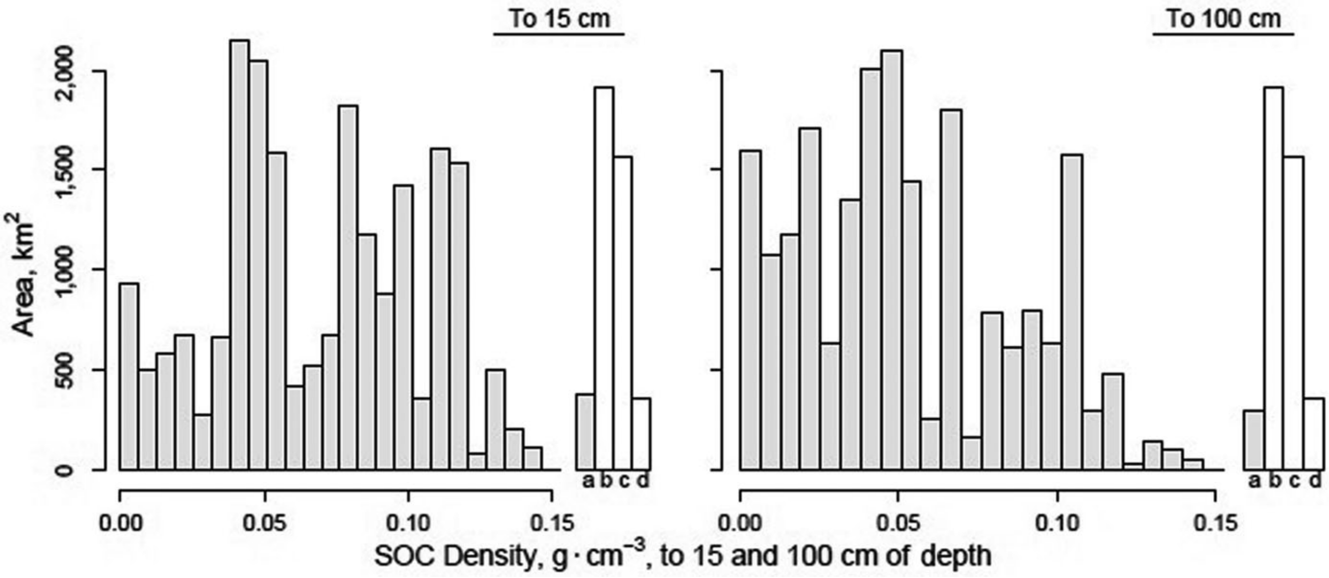

FIGURE 2 Histogram showing the total wetland area for each carbon density across all MU soil types and locations in the upper 15 cm and the upper 100 cm. Bars a–d denote the amount of missing area for special cases: (a) is the excess area-specific SOC densities greater than 0.15 (b) is the total area 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)

When total SOC amount is viewed by coastal region (Table 1), 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 were 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 (Figure 5c). However, the Gulf Coast organic carbon density was the most variant, particularly at shallow depths (Figure 5d).

When viewed by state (Table S2), 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 2, Figure 6, Table S3), 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 U.S. West Coast.

4 | DISCUSSION

4.1 | Summary of wetland blue carbon stocks and fluxes

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

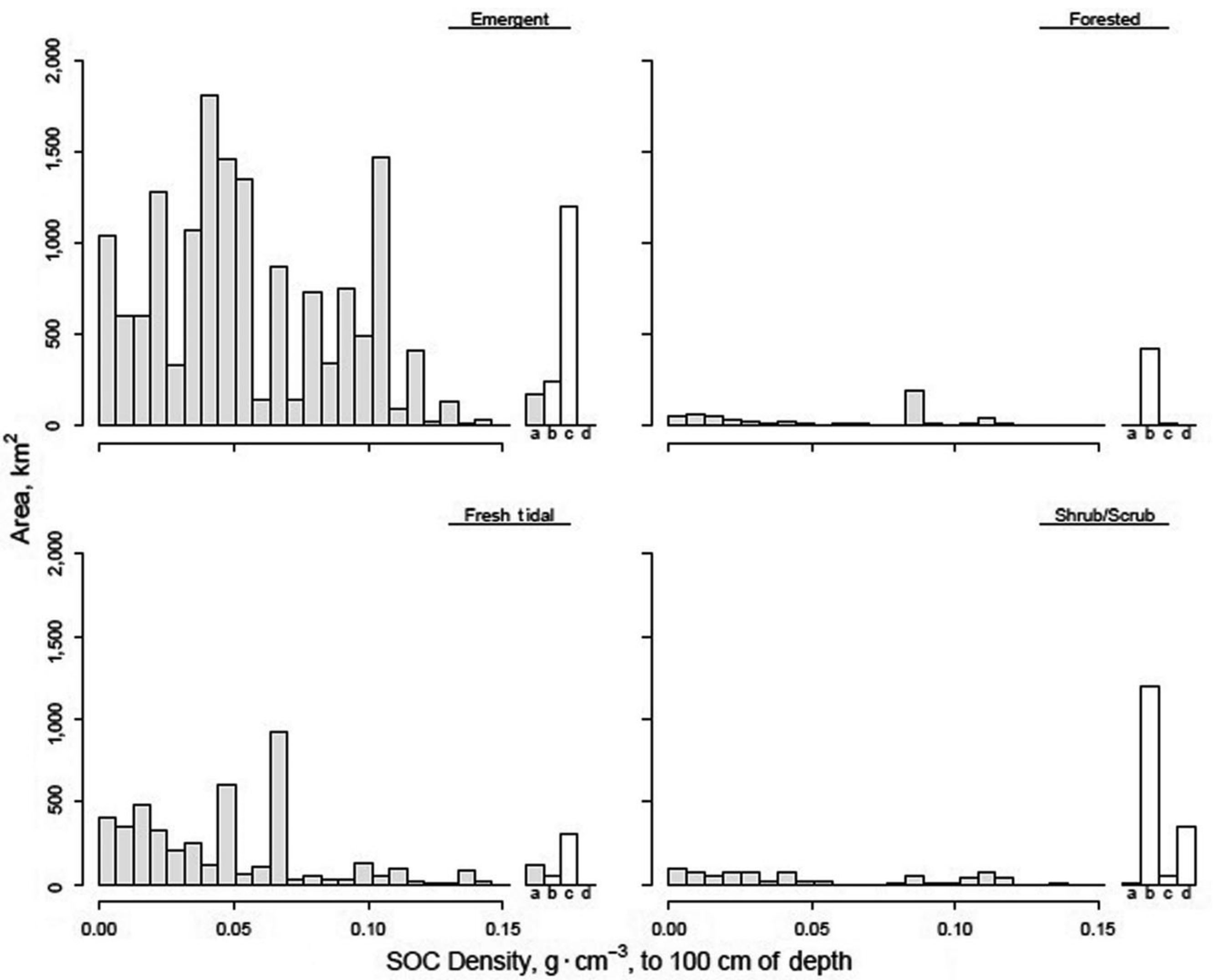

FIGURE 3 Histogram showing the total wetland area for each carbon density across the specific wetland categorizations at a range of 0–100 cm depth interval. In the upper 15 cm, the results are similar and show the same trends as 0–100 cm. Refer to Figure 2 caption for columns a–d

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 (Appendix F of that document), 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, we compared the carbon values from our database versus Ouyang & Lee, 2014 and Chmura et al., 2003 (see Data S1 for more on the differences and comparative benefits of each method).

Based on the stocks database from the current study, we could coarsely estimate carbon flux, where flux refers to the amount of carbon being transferred to or from the wetland with other environments including water, air, and in living organisms, to be ~1.5 Tg C/year for tidal wetland burial in the continental United States. This was obtained by assuming a back-of-the-envelope style calculation of carbon burial rate of 0.006 g cm^{-2} $year^{-1}$, 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 estimate also assumed an average accretion rate of 0.2 cm/year (Callaway et al., 2012; Chmura et al., 2003; Morris et al., 2016). In at least two separate lines of on-going research, we are combining more precise regional sediment accumulation rates with CoBluCarb to create spatially explicit flux rates at finer scales. Though we were unable to find a total flux value for the continental United States from the literature to compare with, it has been estimated that there is ~220 Pg of carbon total and all wetlands (tidal and nontidal) within the United States provide a carbon sink of ~49 Tg C/year, (Bridgham et al., 2006). Globally, mangroves have been estimated to bury ~218 ± 72 Tg C/year (Bouillon et al., 2008). Howard et al. (2017) reported an

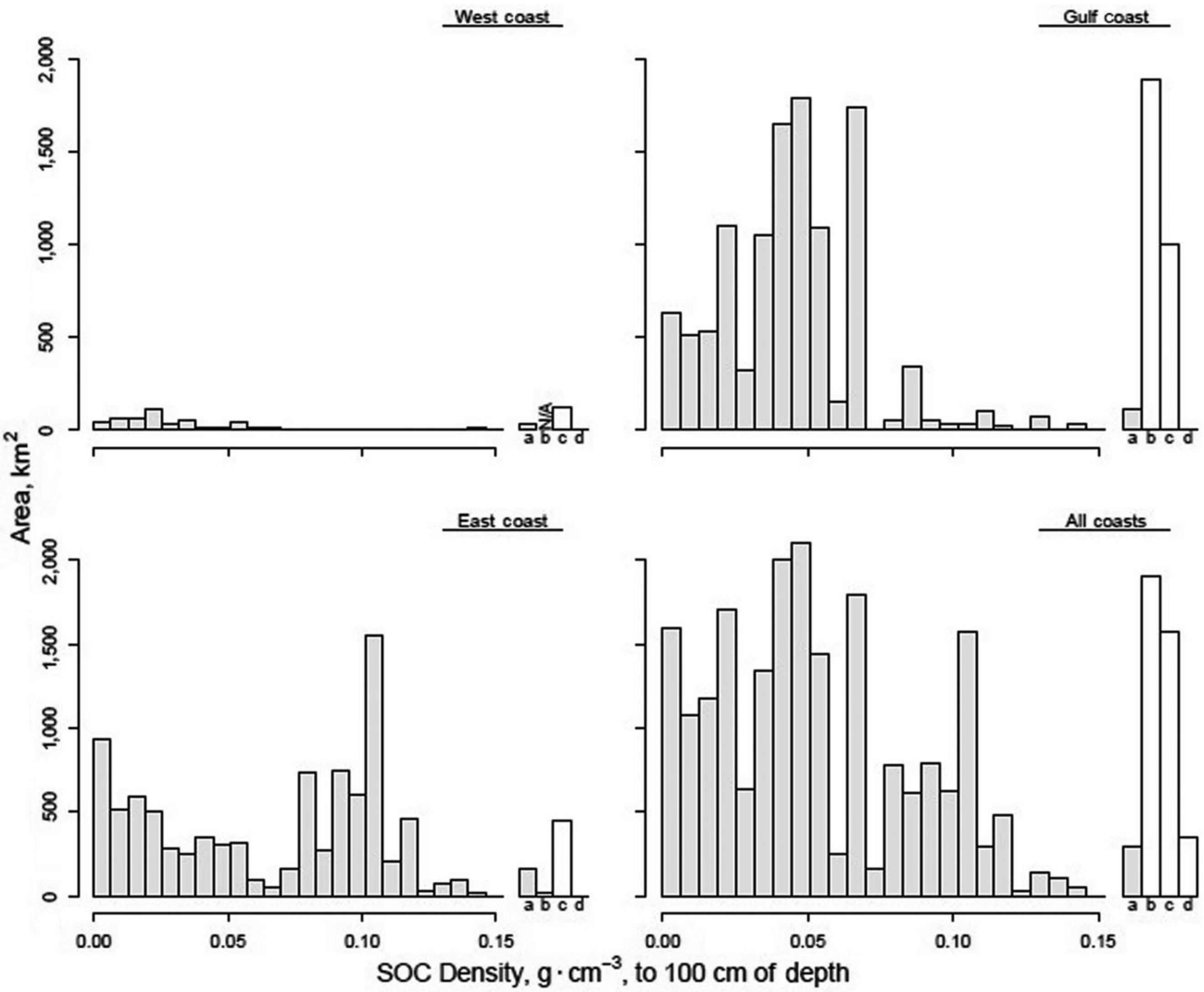

FIGURE 4 Histogram showing the total wetland area for each carbon density across each coast from 0–100 cm depth increments. In the upper 15 cm, the results are similar and show the same trends as 0–100 cm. Refer to Figure 2 caption for columns a–d

estimation range of 10.4–25.1 billion mega grams of carbon in coastal wetlands globally. Chmura et al. (2003) and Duarte et al. (2005) have estimated global burial rates of 4.8 ± 0.5 Tg C/year and 87.2 ± 9.6 Tg C/year 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/year. The spatial distribution of mangroves in certain regions is 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 (Bianchi et al., 2013; Comeaux, Allison, & Bianchi, 2011; Kulawardhana et al., 2015; Osland et al., 2016). 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 tidal wetlands such as salt marshes and tidally influenced forests (Sheehan et al., 2016). For mangroves, there are a multitude of other studies, giving a global carbon burial in Tg C/year 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 U.S. tidal wetlands.

4.2 | Threats facing blue carbon and their effects on current stocks and variation

Much of the mangrove encroachment research has been focused on *Avicennia germinans* in the Gulf and Atlantic Coasts of the United States. Mangrove encroachment has also been well documented, though not as well studied, in Australia (Kelleway et al., 2016). In both these regions, mangrove encroachment will have a defining influence on the future of the ecosystem services provided (Osland et al., 2016). Mangroves can accumulate more belowground carbon or the same amount as salt marshes (Kelleway et al., 2016; Perry & Mendelssohn, 2009). In Louisiana, mangroves replacing salt marshes had no effect on the carbon sequestering properties of the soils,

TABLE 1 Soil organic carbon by wetland type and coast. Lo identifies quantities that were based on polygons for which SSURGO data were 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)
Category	Lo	Hi	Lo	Hi[a]	Lo	Hi[a]	0–15 cm	0–100 cm	At 100 cm
Estuarine Emergent	15,272.5	16,985.4	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	2,073.2	7.9	20.7	40.1	105.5	0.067	0.051	0.033
Freshwater Tidal	4,577.8	4,961.7	43.1	46.7	225.4	244.3	0.063	0.049	0.042
East	9,329.2	9,818.5	113.6	119.6	623.6	656.3	0.081	0.067	0.058
Gulf	11,344.7	14,559.9	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

[a]Refer to table consistency section in Discussion section.

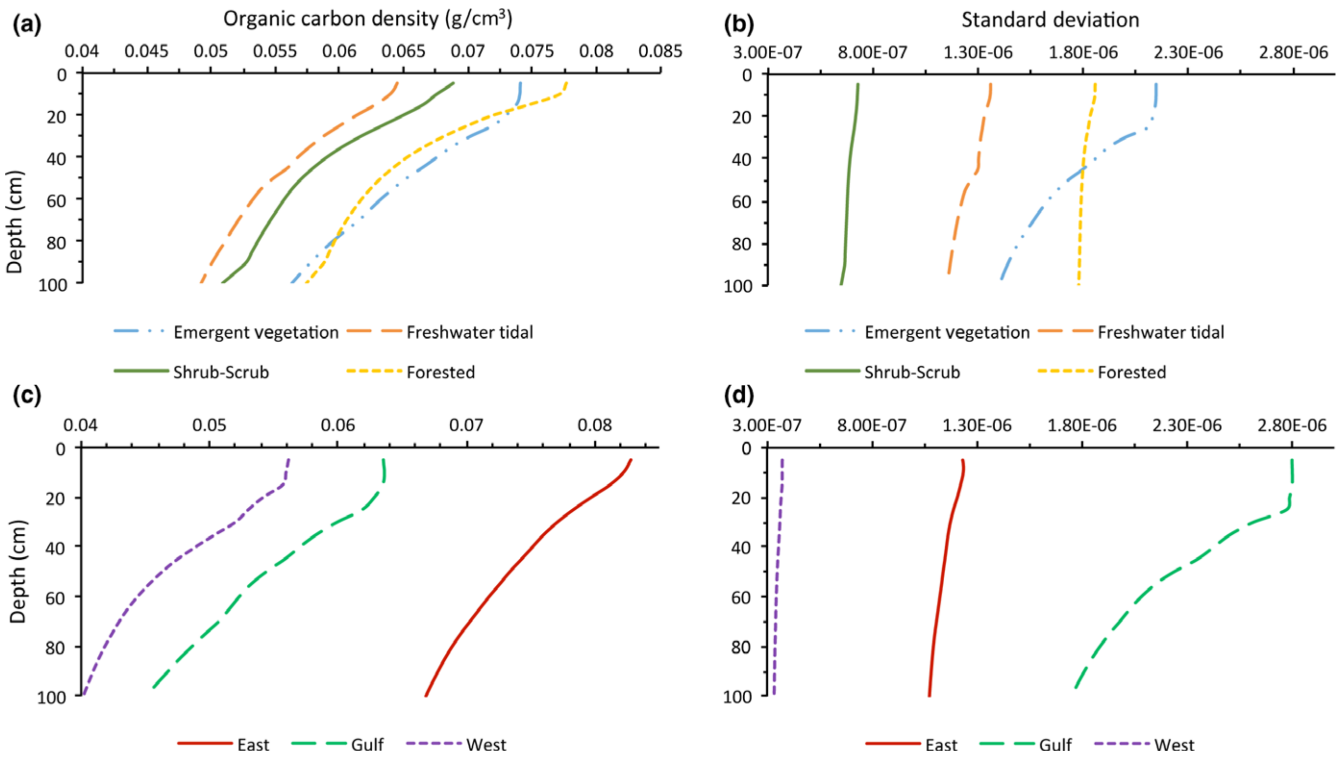

FIGURE 5 Vertical profiles in the upper 100 cm of soil of (a) average area-weighted carbon densities for each wetland type, (b) standard deviation of the average area-weighted carbon density for each wetland type, (c) average area-weighted carbon density for each coast, and (d) standard deviation of the average area-weighted carbon density for each coast [Colour figure can be viewed at wileyonlinelibrary.com]

though due to the currently expanding spatial extent, this may change in the future (Henry & Twilley, 2013; Perry & Mendelssohn, 2009). In the Mississippi Delta, the SOC amount is expected to increase with increased salinity in wetlands, and this may not alter the amount of carbon sequestered, but it will alter the stability of the carbon. (Williams & Rosenheim, 2015). However, in locations in Australia, mangrove encroachment significantly increased belowground C based on peat accumulation, root penetration, and mangrove effects on microbial communities (Kelleway et al., 2016). Parameters such as temperature, presence of plant debris, sea level rise, or rainfall can be the deciding factors for mangrove encroachment, and ultimately drive SOC quantities as well (Gabler et al., 2017; Guo et al., 2017; Osland et al., 2016; Williams & Rosenheim, 2015). In Sanders et al., 2016, it was found that the variability of mangrove carbon stocks was 86% accredited to precipitation. If mangroves and climate change continue to expand as projected, the current SOC densities and carbon stocks in the United States could change drastically (Doughty et al., 2016; Gabler et al., 2017; Yando et al., 2016). In such a scenario, it would be extremely beneficial to have a spatial database such as CoBluCarb. More detailed distinctions between estuarine shrub-scrub and estuarine

TABLE 2 Soil organic carbon by top 15 estuarine drainage areas (EDA), ranked by the 'Hi' stock across 100 cm depth. Lo identifies quantities that were based on polygons for which SSURGO data were 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[a]	Lo	Hi[a]	0–15 cm	0–100 cm	At 100 cm
Atchafalaya/Vermilion Bays	2,282.4	2,465.5	20.8	22.4	116.0	125.3	0.061	0.051	0.042
Chesapeake Bay	1,546.4	1,651.0	17.3	18.4	89.8	95.9	0.074	0.058	0.048
North Ten Thousand Islands	237.6	924.9	3.0	11.6	17.9	69.6	0.084	0.075	0.074
Barataria Bay	750.7	1,151.0	6.7	10.3	37.8	58.0	0.059	0.050	0.033
Breton/Chandeleur Sound	1,120.3	1,247.1	10.0	11.2	48.9	54.4	0.060	0.044	0.025
West Mississippi Sound	728.3	780.3	7.9	8.5	48.9	52.4	0.073	0.067	0.053
Pamlico Sound	634.2	648.5	8.7	8.9	50.0	51.1	0.091	0.079	0.058
St.Catherines/Sapelo Sounds	522.4	532.3	8.4	8.6	49.5	50.4	0.108	0.095	0.087
Terrebonne/Timbalier Bays	1,029.3	1,155.6	7.6	8.6	43.7	49.1	0.049	0.042	0.032
Delaware Bay	673.2	699.0	7.7	8.0	46.9	48.7	0.076	0.070	0.075
St.Andrew/St.Simons Sounds	540.2	551.8	9.7	10.0	45.5	46.5	0.12	0.084	0.076
New Jersey Inland Bays	421.4	431.0	6.0	6.2	39.1	39.9	0.096	0.093	0.095
Albemarle Sound	349.6	357.0	5.2	5.3	33.1	33.8	0.099	0.095	0.073
St.Helena Sound	365.1	379.9	5.5	5.7	31.2	32.5	0.1	0.085	0.074
Mermentau River	735.1	776.6	9.1	9.6	30.5	32.2	0.082	0.041	0.014

[a]Refer to table consistency section in Discussion section.

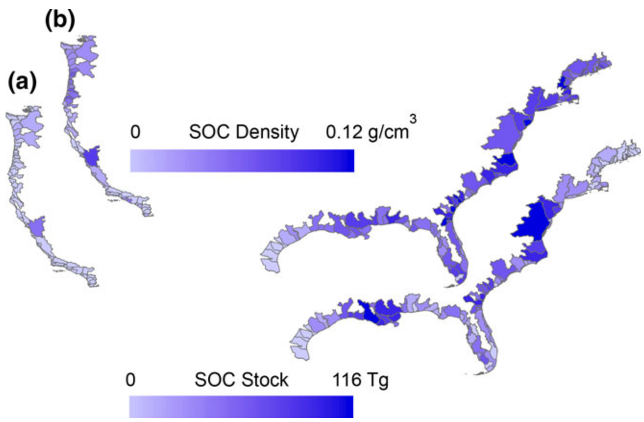

FIGURE 6 (a) Total soil organic carbon across all tidal wetlands in each estuarine basin in the coastal United States (all estuarine basin names listed in Table S2), from 0–100 cm depth, with mapped values based on the minimum range (see text), (b) average SOC density in each estuarine basin, from 0–100 cm depth [Colour figure can be viewed at wileyonlinelibrary.com]

forested wetlands would be helpful to better resolve these changes in CoBluCarb. Future work with CoBluCarb could incorporate other countries help to preserve tidal wetlands in other locations.

4.3 | Accuracy and uncertainty analysis

The strength of our CoBluCarb is in describing the spatial variation of tidal wetland carbon, not necessarily the mean bias (see Data S1 for more on this topic). The area-weighted average of the carbon densities was not highly variable across the wetland types nor the coastal regions, yet given wetland type or coast, the standard deviations were typically vertically arrayed with depth for the wetland types as shown in Figure 5, except for the estuarine 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 (see for example Table S2, and then Table S3 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. For example, comparing the 'hi' estimates in Table 1 for both the coasts and the wetland types. 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 more apparent. All totals and summations for the tables were calculated with the same assumptions and methods for the discrepancies to provide consistency.

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.

4.4 | Utilization and opportunities for CoBluCarb

In estimating the carbon benefits of tidal wetland restoration or conservation projects, we 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, Herr, Laffoley, Tamelander, & Vandever, 2011; Hiraishi et al., 2014; Howard et al., 2014). This depth is used as a default in many greenhouse gas inventories, such as the U.S. National Greenhouse Gas Inventory (EPA 2017) 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 tidal 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, von Unger, Needleman, Crooks, & Emmett-Mattox, 2015; Hiraishi et al., 2014). 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 inconsistencies 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 (Callaway et al., 2012; Craft, Seneca, & Broome, 1991; Keller et al., 2012; 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 U.S.

CoBluCarb can be utilized for many different purposes around the United States, but also for providing an outline for creating similar databases and maps for areas of interest around the world. As part of the Paris Climate Agreement, signatory nations currently look at the U.S. National Gas Inventory efforts as a possible way to achieve Tier 2 accounting of carbon emissions and sequestration potential (EPA 2017). In the 1990s, there were many national and international efforts focused on creating publicly available soil information (Johnston et al., 2003). In this effort, both the Australian Soil Resources Information System (ASRIS) and World Soils and Terrain Digital Database (SOTER) were created along with the United States' SSURGO, STATSGO, and NATSGO (Johnston et al., 2003). In 2013, the first Global Soil Map conference convened, exhibiting results from a mapping project in sub-Saharan Africa (funded by Bill and Melinda Gates Foundation and the Alliance for Green Revolution in Africa [AGRA]) and other worldwide efforts for global soil mapping (Arrouays, McKenzie, Hempel, Forges, & McBratney, 2014). These datasets are extensive and up-to-date, providing spatial datasets similar to the ones used in this study. In cases, such as mangrove encroachment and large scale ecosystem changes, tools such as CoBluCarb can decrease the transition time needed for interpreting the extent of the large scale SOC stock and density changes. This can create an environment in which accurate and influential decisions can be made not only in the best interest of the ecosystems, but for human society as well.

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 do so in the context of both wetland conversion and rising atmospheric CO_2.

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REFERENCES

Amundson, R. (2001). The carbon budget in soils. *Annual Review of Earth and Planetary Sciences*, *29*, 535–562.

Arrouays, D., McKenzie, N., Hempel, J., deForges, A. R., & McBratney, A. B. (2014). *GlobalSoilMap: Basis of the global spatial soil information system*. Boca Raton, FL: CRC Press.

Barbier, E. B., Hacker, S. D., Kennedy, C., Koch, E. W., Stier, A. C., & Silliman, B. R. (2011). The value of estuarine and coastal ecosystem services. *Ecological Monographs*, *81*, 169–193.

Bianchi, T. S., Allison, M. A., Zhao, J., Li, X., Comeaux, R. S., Feagin, R. A., & Kulawardhana, R. W. (2013). Historical reconstruction of mangrove expansion in the Gulf of Mexico: Linking climate change with carbon sequestration in coastal wetlands. *Estuarine Coastal Shelf Science*, *119*, 7–16.

Bouillon, S., Borges, A. V., Castañeda-Moya, E., Diele, K., Dittmar, T., Duke, N. C., … Twilley, R. R. (2008). Mangrove production and carbon sinks: A revision of global budget estimates. *Global Biogeochemical Cycles*, *22*, GB2013.

Breithaupt, J. L., Smoak, J. M., Smith, T. J., Sanders, C. J., & Hoare, A. (2012). Organic carbon burial rates in mangrove sediments: strengthening the global budget. *Global Biogeochemical Cycles*, *26*, GB3011.

Bridgham, S. D., Megonigal, J. P., Keller, J. K., Bliss, N. B., & Trettin, C. (2006). The carbon balance of North American wetlands. *Wetlands*, *26*, 889–916.

Bridgham, S. D., Megonigal, J. P., Keller, J. K., Bliss, N. B., & Trettin, C. (2007). Wetlands. Chapter 13 and Appendix F In A. W. King, L. Dilling, G. Zimmerman, D. Fairman, R. Houghton, G. Marland, A. Rose & T. Wilbanks (Eds.), *The first state of the carbon cycle report (SOCCR): The North American carbon budget and implications for the global carbon cycle* (pp. 139–148). Washington, DC: U.S. Climate Change Science Program, 177-192.

Callaway, J. C., Borgnis, E. L., Turner, R. E., & Milan, C. S. (2012). Carbon sequestration and sediment accretion in San Francisco Bay tidal wetlands. *Estuaries and Coasts*, *35*, 1163–1181.

Chmura, G. L., Anisfeld, S. C., Cahoon, D. R., & Lynch, J. C. (2003). Global carbon sequestration in tidal, saline wetland soils. *Global Biogeochemical Cycles*, *17*, GB001917.

Comeaux, R. S., Allison, M. A., & Bianchi, T. S. (2011). Mangrove expansion in the Gulf of Mexico with climate change: Implications for wetland health and resistance to rising sea-levels. *Estuarine Coastal Shelf Science*, *96*, 81–95.

Costanza, R., d'Arge, R., De Groot, R., Farber, S., Grasso, M., Hannon, B., … van den Belt, M. (1997). The Value of the World's Ecosystem Services and Natural Capital. *Nature*, *387*, 253–260.

Cowardin, L. M., Carter, V., Golet, F. C., & LaRoe, E. T. (1979). Classification of wetlands and deepwater habitats of the United States. *U.S. Fish and Wildlife Service FWS/OBS*, *79*, 131.

Craft, C. B., Seneca, E. D., & Broome, S. W. (1991). Loss on ignition and Kjeldhal digestion for estimating organic carbon and total nitrogen in estuarine marsh soils: Calibration with dry combustion. *Estuaries*, *14*, 175–179.

Crooks, S., Emmett-Mattox, S., & Findsen, J. (2010). Findings of the National Blue Ribbon Panel on the Development of a Greenhouse Gas Offset Protocol for Tidal Wetlands Restoration and Management: action plan to guide protocol development. Findings of the National Blue Ribbon Panel on the Development of a Greenhouse Gas Offset Protocol for Tidal Wetlands Restoration and Management: action plan to guide protocol development, Restore America's Estuaries, Philip Williams & Associates, Ltd., and Science Applications International Corporation.

Crooks, S., Findsen, J., Igusky, K., Orr, M., & Brew, D. (2009). Greenhouse gas mitigation typology issues paper: Tidal wetlands restoration. *California Climate Action Registry*.

Crooks, S., Herr, D., Laffoley, D., Tamelander, J., & Vandever, J. (2011). Regulating Climate Change through Restoration and Management of Coastal Wetlands and Near-shore Marine Ecosystems: Mitigation Potential and Policy Opportunities. *World Bank, IUCN, ESA PWA, Washington, Gland, San Francisco*.

Day, J. W. Jr, Boesch, D. F., Clairain, E. J., Kemp, G. P., Laska, S. B., Mitsch, W. J., … Whigham, D. F. (2007). Restoration of the Mississippi Delta: Lessons from Hurricanes Katrina and Rita. *Science (New York, N.Y.)*, *315*, 1679–1684.

Doughty, C. L., Langley, J. A., Walker, W. S., Feller, I. C., Schaub, R., & Chapman, S. K. (2016). Mangrove range expansion rapidly increases coastal wetland carbon storage. *Estuaries and Coasts*, *39*, 385–396.

Duarte, C. M., Middelburg, J. J., & Caraco, N. (2005). Major role of marine vegetation on the oceanic carbon cycle. *Biogeosciences*, *2*, 1–8.

Emmer, I. E., von Unger, M., Needleman, B., Crooks, S., & Emmett-Mattox, S., (2015). Coastal Blue Carbon in Practice: A Manual for Using the VCS Methodology for Tidal Wetland and Seagrass Restoration. VM0033. *Restore America's Estuaries*.

EPA. (2017). Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2015. Washington, DC: U.S. Environmental Protection Agency.

Federal Geographic Data Committee. (2013). *Classification of wetlands and deep water habitats of the United States, FGDC-STD-004-2013*, Second Edition. Washington, DC: Wetlands Subcommittee, Federal Geographic Data Committee and U.S. Fish and Wildlife Service, 86.

Gabler, C. A., Osland, M. J., Grace, J. B., Stagg, C. L., Day, R. H., Hartley, S. B., … Mcleod, J. L. (2017). Macroclimatic change expected to transform coastal wetland ecosystems this century. *Nature Climate Change*, *7*, 142–147.

Grimsditch, G., Alder, J., Nakamura, T., Kenchington, R., & Tamelander, J. (2013). The blue carbon special edition–Introduction and overview. *Ocean & Coastal Management*, *83*, 1–4.

Guo, H., Weaver, C., Charles, S. P., Whitt, A., Dastidar, S., D'Odorico, P., … Pennings, S. C. (2017). Coastal regime shifts: Rapid responses of coastal wetlands to changes in mangrove cover. *Ecology*, *98*, 762–772.

Henry, K. M., & Twilley, R. R. (2013). Soil Development in a Coastal Louisiana Wetland during a Climate-Induced Vegetation Shift from Salt Marsh to Mangrove. *Journal of Coastal Research*, *29*, 1273–1283.

Hiraishi, T., Krug, T., Tanabe, K., Srivastava, N., Baasansuren, J., Fukuda, M., & Troxler, T. (2014). 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands. IPCC, Switzerland.

Hopkinson, C. S., Cai, W., & Hu, X. (2012). Carbon sequestration in wetland dominated coastal systems—a global sink of rapidly diminishing magnitude. *Current Opinion in Environmental Sustainability*, *4*, 186–194.

Howard, J., Hoyt, S., Isensee, K., Telszewski, M., & Pidgeon, E. (2014). Coastal blue carbon: methods for assessing carbon stocks and emissions factors in mangroves, tidal salt marshes, and seagrasses. Conservation International, Intergovernmental Oceanographic Commission of UNESCO, International Union for Conservation of Nature Arlington, Virginia, USA, 184.

Howard, J., Sutton-Grier, A., Herr, D., Kleypas, J., Landis, E., Mcloed, E., … Simpson, S. (2017). Clarifying the role of coastal and marine systems in climate mitigation. *Frontiers in Ecology and the Environment*, *15*, 42–50.

Jenny, H. (1941). *Factors of soil formation: A system of quantitative pedology*. New York: McGraw-Hill.

Johnston, R., Barry, S., Bleys, E., Bui, E. N., Moran, C. J., Simon, A. D. P., . . . Grundy, M. (2003). ASRIS: The database. *Soil Research*, *41*, 1021–1036.

Keller, J. K., Takagi, K. K., Brown, M. E., Stump, K. N., & Takahashi, C. G. (2012). Soil Organic Carbon Storage in Restored Salt Marshes in Huntington Beach, California. *Bulletin, Southern California Academy of Sciences*, *111*, 153–161.

Kelleway, J. J., Saintilan, N., Macreadie, P. I., Skilbeck, C. G., Zawadzki, A., & Ralph, P. J. (2016). Seventy years of continuous encroachment substantially increases 'blue carbon' capacity as mangroves replace intertidal salt marshes. *Global Change Biology*, *22*, 1097–1109.

Kulawardhana, R. W., Feagin, R. A., Popescu, S. C., Boutton, T. W., Yeager, K. M., & Bianchi, T. S. (2015). The role of elevation and relative sea level history in determining carbon distribution in *Spartina alterniflora* dominated salt marshes. *Estuarine Coastal Shelf Science*, *154*, 48–57.

Laffoley, D., & Grimsditch, G. D. (2009). *The management of natural coastal carbon sinks*. International Union for Conservation of Nature, Gland, Switzerland.

Lau, W. W. (2012). Beyond carbon: Conceptualizing payments for ecosystem services in blue forests on carbon and other marine and coastal ecosystem services. *Ocean & Coastal Management*, *83*, 5–14.

Marland, G., McCarl, B. A., & Schneider, U. (2001). Soil carbon: Policy and economics. *Climatic Change*, *51*, 101–117.

Martínez, M., Intralawan, A., Vázquez, G., Pérez-Maqueo, O., Sutton, P., & Landgrave, R. (2007). The coasts of our world: Ecological, economic and social importance. *Ecological Economics*, *63*, 254–272.

Mcleod, E., Chmura, G. L., Bouillon, S., Salm, R., Bjork, M., Duarte, C. M., . . . Silliman, B. R. (2011). A blueprint for blue carbon: Toward an improved understanding of the role of vegetated coastal habitats in sequestering CO2. *Frontiers in Ecology and the Environment*, *9*, 552–560.

Miles, L., & Kapos, V. (2008). Reducing greenhouse gas emissions from deforestation and forest degradation: Global land-use implications. *Science (New York, N.Y.)*, *320*, 1454–1455.

Morris, J. T., Barber, D. C., Callaway, J. C., Chambers, R., Hagen, S. C., Hopkinson, C. S., . . . Wigand, C. (2016). Contributions of organic and inorganic matter to sediment volume and accretion in tidal wetlands at steady state. *Earth's Future*, *4*, 110–121.

Morris, J. T., Edwards, J., Crooks, S., & Reyes, E. (2012). Assessment of Carbon Sequestration Potential in Coastal Wetlands. Recarbonization of the Biosphere: Ecosystems and the Global Carbon Cycle. *Springer Science*, *24*, 517–531.

Murray, B. C., Pendleton, L., Jenkins, W. A., & Sifleet, S. (2011). Green payments for blue carbon: Economic incentives for protecting threatened coastal habitats. *Nicholas Institute for Environmental Policy Solutions*, Report NI, 11, 04.

Nahik, A. M., & Fennessy, M. S. (2016). Carbon storage in US wetlands. *Nature Communications*, *7*, 13835. https://doi.org/10.1038/ncomms13835

Needelman, B., Crooks, S., Shumway, C., Titus, J., Takacs, R., & Hawkes, J. (2012). *Restore-Adapt-Mitigate: Responding to Climate Change Through Coastal Habitat Restoration* (pp. 1–63). Washington DC: Restore America's Estuaries.

Nellemann, C., Corcoran, E., Duarte, C., Valdés, L., De Young, C., Fonseca, L., & Grimsditch, G. (2009). Blue Carbon: a rapid response assessment. United Nations Environment Programme, GRID-Arendal, 78.

Osland, M. J., Enwright, N. M., Day, R. H., Gabler, C. A., Stagg, C. L., & Grace, J. B. (2016). Beyond just sea-level rise: Considering macroclimatic drivers within coastal wetland vulnerability assessments to climate change. *Global Change Biology*, *22*, 1–11.

Ouyang, X., & Lee, S. (2014). Updated estimates of carbon accumulation rates in coastal marsh sediments. *Biogeosciences*, *11*, 5057–5071.

Pendleton, L., Donato, D. C., Murray, B. C., Crooks, S., Jenkins, W. A., Sifleet, S., . . . Baldera, A. (2012). Estimating global "blue carbon" emissions from conversion and degradation of vegetated coastal ecosystems. *PLoS ONE*, *7*, e43542.

Perry, C. L., & Mendelssohn, I. A. (2009). Ecosystem effects of expanding populations of *Avicennia germinans* in a Louisiana salt marsh. *Wetlands*, *29*, 396–406.

Peters-Stanley, M., Hamilton, K., Marcello, T., Orejas, R., Thiel, A., & Yin, D. (2012). Developing dimension: state of the voluntary carbon markets. Ecosystem marketplace & Bloomberg new energy finance, New York, New York.

Pribyl, D. W. (2010). A critical review of the conventional SOC to SOM conversion factor. *Geoderma*, *156*, 75–83.

Sanders, C. J., Maher, D. T., Tait, D. R., Williams, D., Holloway, C., Sippo, J. Z., & Santos, I. R. (2016). Are global mangrove carbon stocks driven by rainfall? *Journal of Geophysical Research: Biogeosciences*, *121*, 2600–2609.

Sheehan, L., Crooks, S., Tomasko, D., Robison, D., Connell, H., & Quinton, B. (2016). Tampa Bay Blue Carbon Assessment: Summary of Findings. Report to Restore America's Estuaries and the Tampa Bay Estuary Program.

Soil Survey Division Staff. (1993). *Soil survey manual*. Washington, DC, USA: United States Department of Agriculture, 18.

Ullman, R., Bilbao-Bastida, V., & Grimsditch, G. (2012). Including blue carbon in climate market mechanisms. *Ocean & Coastal Management*, *83*, 15–18.

Williams, K. E., & Rosenheim, B. E. (2015). What happens to soil organic carbon as coastal marsh ecosystems change in response to increasing salinity? An exploration using ramped pyrolysis. *Geochemistry, Geophysics, Geosystems*, *16*, 2322–2335.

Wylie, L., Sutton-Grier, A. E., & Moore, A. (2016). Keys to successful blue carbon projects: Lessons learned from global case studies. *Marine Policy*, *65*, 76–84.

Yando, E. S., Osland, M. J., Willis, J. M., Day, R. H., Krauss, K. W., & Hester, M. W. (2016). Salt marsh-mangrove ecotones: Using structural gradients to investigate the effects of woody plant encroachment on plant-soil interactions and ecosystem carbon pools. *Journal of Ecology*, *104*, 1020–1031.

Zhong, B., & Xu, Y. (2011). Scale effects of geographical soil datasets on soil carbon estimation in Louisiana, USA: A comparison of STATSGO and SSURGO. *Pedosphere*, *21*, 491–501.

SUPPORTING INFORMATION

Additional Supporting Information may be found online in the supporting information tab for this article.

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