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Does Regional Development Influence Sedimentary Blue Carbon Stocks? A Case Study From Three Australian Estuaries

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Mitigating climate change through the reduction of atmospheric CO_2 levels is of interest, particularly through maintaining and re-establishing natural ecosystems that act as carbon sinks, such as coastal vegetated habitats or "blue carbon" systems. Here we compare sedimentary blue carbon (C) stocks from 37 sediment cores collected in pristine (n = 13), agricultural (n = 11), and urban (n = 13) estuaries within the same geomorphological region, located on the eastern coast of Australia. The mean estimated C stocks for each carbon system (seagrass, mangrove, and saltmarshes) were 402 ± 78, 830 ± 109, and 723 ± 100 Mg C ha^{-1}, respectively, conservatively estimated up to 3 m depths. Analysis of variance revealed no significant difference between C stocks per area (C ha^{-1}) considering each habitat type and between specific estuaries. However, the total estuarine C stocks were found to be greater with increasing levels of conservation, based on larger areas of blue carbon vegetation. The potential loss of C to the atmosphere from these small regional estuaries are 500,574 ± 118,635 tons of CO_2 equivalent (CO_2e), based on specific assumptions. The implication of these results are that there are large C stocks in small regional estuaries which supports the protection of blue C systems in developing coastal areas and highlights the uncertainties of the CO_2 emissions from potential blue C habitat degradation.

Keywords: soil carbon stocks, seagrass, mangrove, saltmarsh, estuaries, Australia

INTRODUCTION

Vegetated coastal ecosystems (i.e., seagrass beds, saltmarsh meadows, and mangrove forests) are highly productive ecosystems that play a critical role in the global carbon, water, and nutrient cycles (Sanders et al., 2014; Lovelock et al., 2017; Maher et al., 2018). Their natural ability to sequester substantial amounts of carbon dioxide (CO_2) from the atmosphere via the long term burial of carbon in mostly anoxic sediments has become increasingly recognized as important climate change mitigation strategies (Chmura et al., 2003; McLeod et al., 2011; Hopkinson et al., 2012;

Howard et al., 2017; Macreadie et al., 2017a). However, while these blue C systems are capable of offsetting anthropogenic greenhouse gas emissions through bio- sequestering atmospheric CO_2, their continuing decline due to environmental change and human land use activities is reducing their capacity to provide this ecosystem service and ultimately exacerbating global climate change (McLeod et al., 2011; Beaumont et al., 2014; Atwood et al., 2017).

Over the past several decades, it is estimated that about one third of global mangroves, seagrass, and saltmarsh have been lost as a result of deforestation and habitat reclamation, coastal and urban development, nutrient enrichment, water quality degradation, and climate change (Alongi, 2002; Orth et al., 2006; Gedan et al., 2009; McLeod et al., 2011; Pendleton et al., 2012; Lovelock et al., 2015). This decline in coastal blue C systems still continues today at rates estimated between ~0.5–3% annually depending on ecosystem type (McLeod et al., 2011; Pendleton et al., 2012). The degradation of these habitats, particularly the disturbance of sediments, leads to the remineralization of CO_2 that has been stored for millennia (McLeod et al., 2011; Lovelock et al., 2017). An evaluation of global blue carbon emissions emitted as a result of the conversion and degradation of vegetated coastal ecosystems has been estimated at 0.15–1.02 Pg (billion tons) of CO_2 being released annually which is estimated to cause economic damages of approximately $US 6–42 billion annually (Pendleton et al., 2012). These economic damages are associated with restoration efforts in wetlands with ecological benefits and undefined carbon offsets (Irving et al., 2011).

Although there has been an increasing amount of studies focusing on quantifying potential CO_2 emissions from blue carbon sediment disturbance to evaluate the effects of habitat loss on greenhouse gas budgets (Pendleton et al., 2012; Brown et al., 2016; Atwood et al., 2017; Lovelock et al., 2017; Macreadie et al., 2017b), there are still uncertainties when it comes to variabilities in C stock estimates, particularly in regards to inter-ecosystem variability (Lavery et al., 2013; Friess and Webb, 2014; Ricart et al., 2015; Hayes et al., 2017; Kelleway et al., 2017). While variations in sedimentary carbon stocks in blue C systems have been documented across longitudinal and estuarine spatial gradients (Lavery et al., 2013; Brown et al., 2016; Sanders et al., 2016; Hayes et al., 2017; Lewis et al., 2017), data on the variability of C stocks from blue C systems of the same geomorphological region are scarce.

The current study was therefore undertaken with the objective of (i) estimating and evaluating variability in sedimentary blue carbon stocks of three estuaries within the same geomorphological region and (ii) quantifying possible variability in sedimentary C stocks as a result of regional development. A total of 13 sedimentary C stock estimates were estimated from sediment cores collected from blue C systems within a relatively pristine estuarine catchment, 11 cores from a predominately agricultural impacted estuarine catchment, and 13 sediment cores were collected from an urbanized estuary catchment. For comparative purposes, extrapolations of saltmarsh, mangrove and seagrass sediment C stocks were calculated for each estuary and the potential CO_2 emissions from habitat degradation were based on these sedimentary C stock estimates.

MATERIALS AND METHODS

Study Sites

Field observations were performed along three coastal estuaries on the subtropical East Coast of New South Wales, Australia (**Figure 1**). The Wooli River, Corindi River, and Coffs Creek estuaries are all tidal estuaries situated within the same geomorphological region (within 50 km of each other) and experience a similar subtropical climate of hot wet summers and cold dry winters. The region receives an average annual rainfall of approximately 1600 mm with the wettest and driest months being February and September, respectively. Each estuary contains all three blue C systems with the dominant species for each blue carbon system being *Zostera muelleri* (seagrass), *Sporobolus virginicus* (saltmarsh), and *Avicennia marina* (mangrove).

Although the three estuaries in this study experience a similar climate, they differ in terms of the degree of environmental impact and regional development within their catchments. The estuary situated the furthest north of all estuaries, the Wooli River estuary, is located in the Yuraygir National Park in the Northern Rivers district of New South Wales, Australia (29.89° S, 153.27° E) (**Figure 1**). The closest villages are Minnie Water, situated at 13.7 km north from the Wooli River, and Red Rock, which is found 48 km south from Wooli. Although in the surroundings of the river there are camping and caravan parks, as well as holiday apartments, the Wooli River is known to be one of the most pristine systems in New South Wales, Australia. The Wooli River estuary has an area of 370 ha and a total catchment area of 18,000 ha.

Also situated in the Northern Rivers district of New South Wales, the Corindi River estuary (29.98° S, 153.23° E) and its surrounding catchment (190 ha) drains to the ocean in the town of Red Rock, NSW (population: 435, ABS, 2016; **Figure 1**). The catchment area is primarily dominated by agricultural development and supports a range of blueberry and banana plantations. The Corindi River drainage basin is one of the major blueberry growing areas in northern New South Wales and previous studies have shown the impacts of blueberry farms to the local estuary, including phosphorus enrichment (Conrad et al., 2018).

The urban estuary studied in this work, Coffs Creek estuary (30.30° S, 153.14° E), is situated in the northern section of the Mid North Coast near the township of Coffs Harbour, NSW Australia (**Figure 1**). The estuary has a catchment area of about 250 ha of which 80% is dominated by urban development and agriculture, while just 16% is considered undisturbed (Roper et al., 2011). Coffs Creek estuary has an estimated population of ~18,000 (Ryder et al., 2012). The water demand of Coffs Harbour is around 18 ML per day. Potable water for the city is pumped from the local rivers outside the catchment (Orara River, Nymboida River,

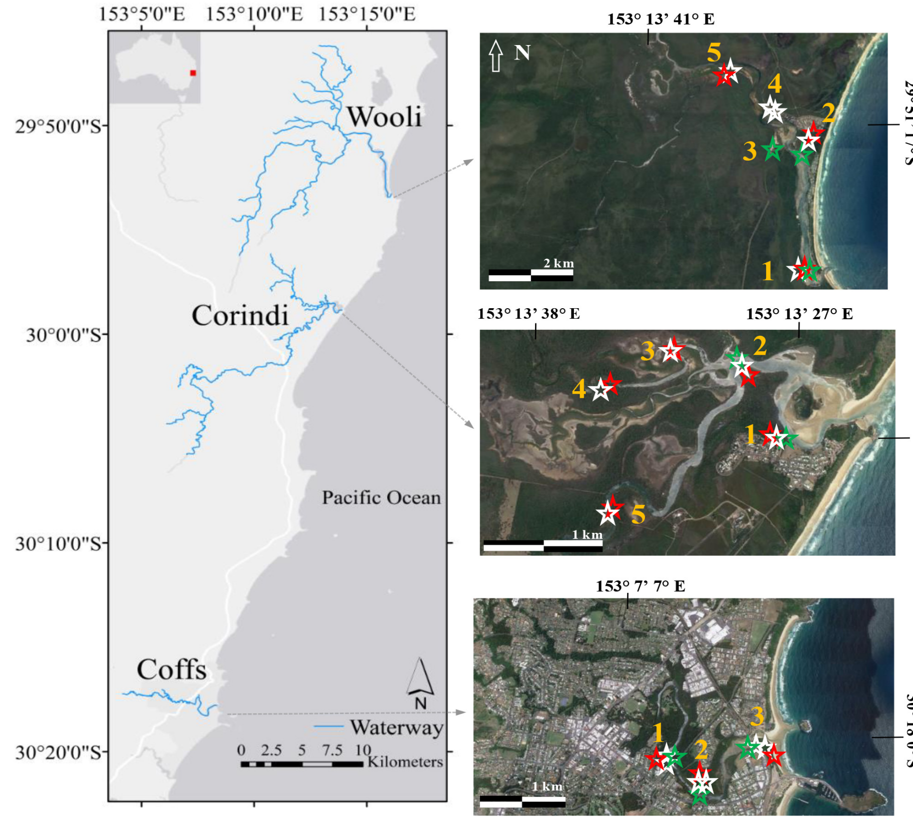

FIGURE 1 | Study region on the left, and satellite image of each estuary on the right. The townships of Coffs Harbour, Corindi Beach and Wooli, NSW [populations: 72,971, 1,200, and 418, respectively (ABS, 2016)]. Coffs Harbour is the main urban center of the region. Numbers on the satellite image of each area indicate location of sediment samples, greens stars represent seagrass, white stars mangrove and red stars saltmarsh sediment cores.

and Shannon Creek) and the wastewater from the city is treated before the effluent is disposed offshore (Ryder et al., 2012).

Sample Collection and C Stock Calculations

Sampling was conducted in March 2016 (Coffs Creek estuary), March 2017 (Corindi River estuary), and March 2018 (Wooli River estuary). Using a 50 cm long, 5 cm diameter Russian Peat auger (mangrove and saltmarsh) or PVC corer with a 7 cm inner diameter (seagrass). Sediment cores were collected, at least two sediment cores in each blue carbon system, along each estuary (**Figure 1**). At each sampling site, soil depth was measured using a 3 m metal rod to probe for soil bedrock. If the rod became fully submerged in the sediment, a conservative depth of 3 m was presumed. Once collected, sediment cores were sectioned into 2 cm intervals. This sampling protocol is designed to allow maximum spatial coverage, based on available resources, and allowed for comparisons with other blue carbon stock studies that sampled three depth intervals per sediment core (Lewis et al., 2017).

Dry bulk density (DBD) was calculated for each interval using the dry weight of sediment and volume of sample from the sediment core. Three subsamples from each core (8–10, 28–30, and 48–50 cm) were placed into the oven at 105°C until dry, to obtain the sample dry weight. The three samples per sediment core were then placed in the furnace at 550°C to burn off the organic matter [loss on ignition (L.O.I.)] (Sanders et al., 2012). Organic C content was calculated by multiplying the organic material, determined from the loss on ignition (LOI) method, by 0.58 as confirmed by Brown et al. (2016) for this region. By multiplying soil depth, DBD and organic C content, sediment C stocks for each core were estimated. To determine the C stocks in saltmarsh, mangrove and seagrass sediments of each estuary, the mean C stocks were multiplied by the area of each habitat (**Table 1**).

TABLE 1 | Total area (ha) of each blue carbon ecosystem within Wooli River estuary, Corindi River estuary, and Coffs Creek estuary.

	Wooli River (ha)	Corindi River (ha)	Coffs Creek (ha)	NSW (ha)
Seagrass	9.4	2.4	0.2	15, 900
Mangrove	86.0	37.1	20.1	12, 500
Saltmarsh	66.9	52.7	1.4	7, 200

Data Analysis

To evaluate differences in carbon stocks (Mg C ha^{-1}), carbon density (g C cm^{-3}), and carbon content (g C g $sediment^{-1}$) between locations (Coffs, Corindi, and Wooli; fixed factor), habitat types (seagrass, mangrove, and saltmarsh; fixed factor), and cores (three replicate subsamples; random, nested within location and habitat type), we ran a distance-based multiple factor PERMANOVA on a Euclidian similarity matrix of all samples. We used 9999 permutations of residuals under a reduced model to generate *P*-values and Type III (partial) sums of squares to account for the unbalanced number of cores in each location. The spatial distribution of carbon stock, carbon density, and carbon content was visualized using a non-metric MDS (multidimensional scaling) scatter plot. All analyses were done using PRIMER 7 (Clarke and Gorley, 2015) and PERMANOVA+ (Anderson, 2008).

RESULTS

Below we outline the results for each blue C systems from the Wooli River, Corindi River and Coffs Creek estuaries individually. Sedimentary blue carbon stock data for Coffs Creek estuary are summarized from Brown et al. (2016). All confidence intervals are given as Standard Error. See **Supplementary Table 1** for raw data.

Wooli River Estuary

Seagrass

Carbon content (percentage C of total sample dry weight) in the seagrass sediment samples from the Wooli River estuary ranged from 0.75 to 0.96% with an average of 0.85 ± 0.11% (**Table 2**). The dry bulk densities (DBD) ranged from 1.48 to 1.55 g cm^{-3} with an average DBD of 1.51 ± 0.06 g cm^{-3}. From these, sedimentary seagrass carbon stocks were estimated between 348 and 391 Mg C ha^{-1}, with an overall average of 371 ± 31 Mg C ha^{-1}.

Mangrove

Carbon content in the mangrove sediment samples from the Wooli River estuary ranged from 1.45 to 5.11% with an average of 2.76 ± 0.37% (**Table 2**). The DBD ranged from 0.86 to 1.09 g cm^{-3} with an average DBD of 0.96 ± 0.05 g cm^{-3}. From these, sedimentary mangrove carbon stocks were estimated between 187 and 1332 Mg C ha^{-1}, with an overall average of 642 ± 94 Mg C ha^{-1}.

Saltmarsh

Carbon content in the saltmarsh sediment samples from the Wooli River estuary ranged from 3.78 to 8.42% with an average of 5.91 ± 1.01% (**Table 2**). The DBD ranged from 0.48 to 1.32 g cm^{-3} with an average DBD of 1.02 ± 0.15 g cm^{-3}. From these, sedimentary saltmarsh carbon stocks were estimated between 678 and 1326 Mg C ha^{-1}, with an overall average of 898 ± 172 Mg C ha^{-1}.

Corindi River Estuary

Seagrass

Carbon content in the seagrass sediment samples ranged from 0.78 to 2.33% with an average of 1.55 ± 0.77% (**Table 2**). The DBD ranged from 1.18 to 1.33 g cm^{-3} with an average DBD of 1.26 ± 0.06 g cm^{-3}. From these, sedimentary seagrass carbon stocks were estimated between 102 and 944 Mg C ha^{-1}, with an average of 523 ± 338 Mg C ha^{-1}.

Mangrove

Carbon content in the mangrove sediment samples ranged from 2.04 to 5.86% with an average of 3.81 ± 0.43% (**Table 2**). The DBD ranged from 0.59 to 1.00 g cm^{-3} with an average DBD of 0.79 ± 0.05 g cm^{-3}. From these, sedimentary mangrove carbon stocks were estimated between 469 and 1439 Mg C ha^{-1} with an average of 739 ± 101 Mg C ha^{-1}.

Saltmarsh

Carbon content in the saltmarsh sediment samples ranged from 1.38 to 20.62% with an average of 5.65 ± 2.80% (**Table 2**). The DBD ranged from 0.45 to 1.06 g cm^{-3} with an average DBD of 0.87 ± 0.08 g cm^{-3}. From these, sedimentary saltmarsh carbon stock estimates were estimated between 150 and 1102 Mg C ha^{-1} with an average of 445 ± 123 Mg C ha^{-1}.

Coffs Creek Estuary

Seagrass

Carbon content in the seagrass sediment samples ranged from 0.22 to 2.28% with an average of 1.26 ± 0.48% (**Table 2**). The DBD ranged from 1.18 to 1.86 g cm^{-3} with an average DBD of 1.47 ± 0.15 g cm^{-3}. From these, sedimentary seagrass carbon stocks were estimated between 126 and 589 Mg C ha^{-1}, with an average of 365 ± 113 Mg C ha^{-1}.

Mangrove

Carbon content in the mangrove sediment samples ranged from 1.23 to 9.70% with an average of 4.17 ± 1.29% (**Table 2**). The DBD ranged from 0.55 to 1.52 g cm^{-3} with an average DBD of 0.95 ± 0.14 g cm^{-3}. From these, sedimentary mangrove carbon stocks were estimated between 283 and 2191 Mg C ha^{-1} with an average of 1070 ± 350 Mg C ha^{-1}.

Saltmarsh

Carbon content in the saltmarsh sediment samples ranged from 4.67 to 6.97% with an average of 5.62 ± 1.05% (**Table 2**). The DBD ranged from 0.62 to 1.02 g cm^{-3} with an average DBD of 0.81 ± 0.12 g cm^{-3}. From these, sedimentary saltmarsh carbon stock estimates were estimated between 740 and 1112 Mg C ha^{-1} with an average of 904 ± 109 Mg C ha^{-1}.

Statistical Analysis

Despite the variabilities in sedimentary C stocks observed within and between estuaries (**Table 3**), the results of the PERMANOVA revealed no significant interaction between C stock per unit area, C density or C content between locations (p = 0.46, *pseudo-F* = 0.79), habitats (p = 0.15, *pseudo-F* = 2.10), or habitat amongst

TABLE 2 | Depth (cm), C content (%), dry bulk density (g cm^{-3}), and C stocks (Mg C ha^{-1}) for seagrass, mangrove, and saltmarsh cores collected from Wooli and Corindi River estuaries.

Study site	Habitat and location	Depth (cm)	Carbon content (%)	Dry bulk density (g cm^{-3})	Carbon stock (Mg C ha^{-1})
Wooli River	Seagrass site 1	300	0.96 ± 0.37	1.48 ± 0.17	390.65 ± 98.58
	Seagrass site 2	300	0.75 ± 0.02	1.55 ± 0.12	348.09 ± 34.76
	Seagrass site 3	300	0.84 ± 0.04	1.48 ± 0.05	374.83 ± 11.46
	Seagrass average	**300 ± 0**	**0.85 ± 0.11**	**1.51 ± 0.06**	**371.19 ± 30.99**
	Mangrove site 1	122	1.45 ± 0.51	1.09 ± 0.06	186.50 ± 58.57
	Mangrove site 3	219	2.98 ± 1.10	0.96 ± 0.15	560.87 ± 161.96
	Mangrove site 4R	300	2.10 ± 0.41	1.00 ± 0.05	641.46 ± 158.58
	Mangrove site 4A	300	2.18 ± 0.23	0.86 ± 0.10	545.61 ± 4.96
	Mangrove site 5	300	5.11 ± 0.84	0.90 ± 0.17	1332.22 ± 185.45
	Mangrove average	**248 ± 18**	**2.76 ± 0.37**	**0.96 ± 0.05**	**642.64 ± 93.61**
	Saltmarsh site 1	97	5.53 ± 1.89	1.26 ± 0.07	690.86 ± 264.73
	Saltmarsh site 3	166	8.42 ± 1.18	0.48 ± 0.03	678.32 ± 121.73
	Saltmarsh site 5	300	3.78 ± 1.33	1.32 ± 0.24	1326.01 ± 366.60
	Saltmarsh average	**187 ± 28**	**5.91 ± 1.01**	**1.02 ± 0.15**	**898.39 ± 172.36**
Coridni River	Seagrass site 1	110	0.78 ± 0.06	1.18 ± 0.08	101.74 ± 13.52
	Seagrass site 2	300	2.33 ± 1.53	1.33 ± 0.08	944.34 ± 628.42
	Seagrass average	**205 ± 95**	**1.55 ± 0.77**	**1.26 ± 0.06**	**523.04 ± 338.40**
	Mangrove site 1	300	4.09 ± 0.73	0.59 ± 0.05	740.90 ± 175.65
	Mangrove site 2	300	5.86 ± 0.93	0.81 ± 0.07	1438.85 ± 319.16
	Mangrove site 3	300	2.04 ± 0.26	1.00 ± 0.12	603.78 ± 67.12
	Mangrove site 4	135	4.42 ± 0.07	0.82 ± 0.11	490.02 ± 57.17
	Mangrove site 5 (upper tidal)	240	3.69 ± 1.65	0.88 ± 0.15	689.32 ± 219.60
	Mangrove site 5 (lower tidal)	245	2.64 ± 0.66	0.75 ± 0.06	469.20 ± 79.14
	Mangrove average	**253 ± 26**	**3.81 ± 0.43**	**0.79 ± 0.05**	**738.72 ± 100.74**
	Saltmarsh site 1	300	20.62 ± 11.45	0.45 ± 0.25	1102.40 ± 470.78
	Saltmarsh site 2	170	1.43 ± 0.16	0.98 ± 0.04	242.28 ± 46.84
	Saltmarsh site 3	300	1.38 ± 0.33	0.95 ± 0.13	386.67 ± 86.02
	Saltmarsh site 4	150	3.15 ± 1.63	0.90 ± 0.19	345.90 ± 104.67
	Saltmarsh site 5	85	1.67 ± 0.35	1.06 ± 0.07	149.65 ± 31.05
	Saltmarsh average	**201 ± 43**	**5.65 ± 2.80**	**0.87 ± 0.08**	**445.38 ± 123.06**
Coffs Creek	Seagrass site 1 (middle of patch)	228	2.28 ± 0.56	1.18 ± 0.11	588.65
	Seagrass site 1 (middle of patch)	228	1.81 ± 0.18	1.27 ± 0.06	522.49
	Seagrass site 2	210	0.73 ± 0.47	1.56 ± 0.17	223.26
	Seagrass site 3	300	0.22 ± 0.05	1.86 ± 0.12	125.94
	Seagrass average	**242 ± 20**	**1.26 ± 0.48**	**1.47 ± 0.15**	**365.09 ± 112.57**
	Mangrove site 1 (upper tidal)	300	1.33 ± 0.52	0.71 ± 0.13	282.72 ± 134.96
	Mangrove site 1 (lower tidal)	300	5.20 ± 0.95	0.55 ± 0.10	802.18 ± 27.26
	Mangrove site 2 (upper tidal)	300	4.46 ± 2.65	1.52 ± 0.08	2104.16 ± 1272.18
	Mangrove site 2 (lower tidal)	300	1.23 ± 0.36	0.99 ± 0.33	354.54 ± 186.31
	Mangrove site 3 (upper tidal)	300	3.07 ± 1.37	0.92 ± 0.20	684.27 ± 110.71
	Mangrove site 3 (lower tidal)	300	9.70 ± 5.38	1.03 ± 0.29	2191.00 ± 571.18
	Mangrove average	**300 ± 0**	**4.16 ± 1.29**	**0.95 ± 0.14**	**1069.81 ± 350.16**
	Saltmarsh site 1	248	4.67 ± 1.67	1.02 ± 0.24	1111.87 ± 342.21
	Saltmarsh site 2	180	6.97 ± 2.34	0.78 ± 0.25	861.00 ± 332.54
	Saltmarsh site 3	300	5.22 ± 1.94	0.62 ± 0.10	740.18 ± 0.95
	Saltmarsh average	**243 ± 35**	**5.62 ± 1.05**	**0.81 ± 0.12**	**904.35 ± 109.47**

Coffs Creek estuary data can be found in Brown et al. (2016). Dry bulk density, C content, and C stock data are means. Errors refer to standard errors. The bold values are averages.

locations (p = 0.55, *pseudo-F* = 0.77). There was however, a significant difference among replicates (cores) within each habitat and location (p = 0.004, *pseudo-F* = 2.30) (**Table 3**). Differences in C stock, C density and C content among cores within the same habitat and location were responsible for the greatest components of variation (**Table 3**). All results however, must be interpreted with caution as sample size within locations and habitats was relatively low.

TABLE 3 | Estimates of components of variation from PERMANOVA.

Source	Estimate of component of variation	Sq. root
Habitat	22325	149.42
Location	−4395.4	−66.298
Habitat × location	−13975	−118.22
Replicates	1.2956E+05	359.94
Residual	2.9818E+05	546.06

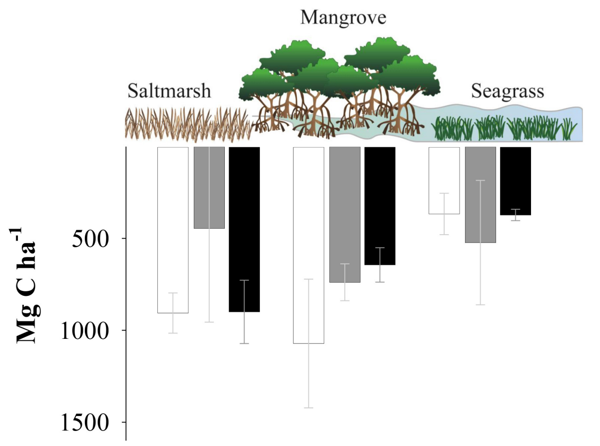

FIGURE 2 | Carbon stocks (Mg C ha^{-1}) of the urbanized Coffs Creek estuary (white), the agriculturally impacted Corindi River estuary (gray) and the pristine Wooli River estuary (black). Data are presented in means while bars represent standard errors. Analysis of variance found no significant differences between habitat type or location.

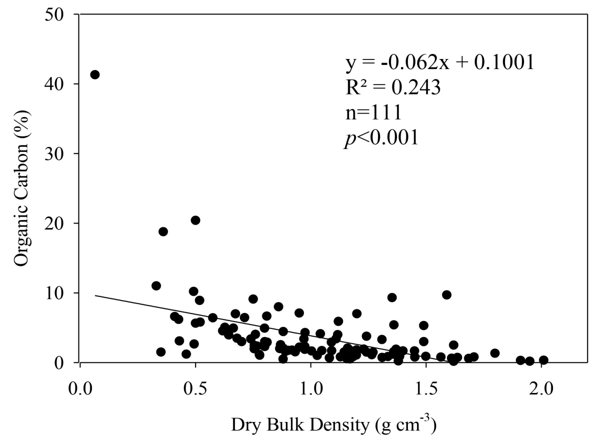

FIGURE 3 | Negative relation between organic carbon content (%) and dry bulk density (g cm^{-3} year^{-1}) from the sediment core intervals of this work.

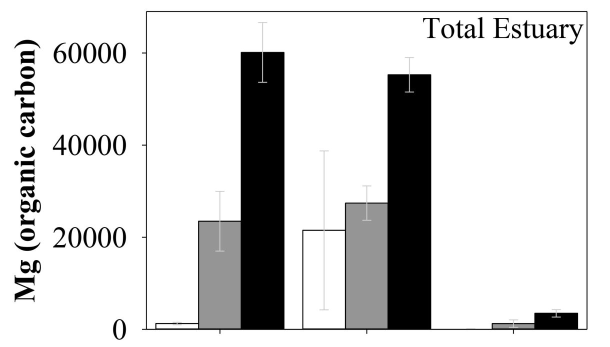

FIGURE 4 | Total carbon stock estimates for seagrass, mangrove and saltmarsh in Coffs Creek (white), Corindi estuary (gray), and Wooli estuary (black) (Mg C). Error bars indicate the standard error.

DISCUSSION

Variabilities in Sedimentary Blue Carbon Stocks

Global averages indicate that mangroves systems have generally higher sedimentary organic C densities than any other blue carbon system (Chmura et al., 2003). However, the results of this study indicate no significant difference in sedimentary C stocks per area (based on the top 3 m of sediment) between all three studied blue C systems (**Figure 2**). Global distribution of blue C systems is primarily governed by physical tolerances to climatic conditions such as temperature and rainfall, which limits and restricts mangrove and seagrass production toward temperate climates (Duarte et al., 2013), while saltmarsh are generally more adapted to cooler climates (Chmura et al., 2003). Given that all three estuaries are within close proximity and are all located in the subtropical climatic region, the climatic conditions may be favorable for saltmarsh production and less favorable for mangrove and seagrass, resulting in similar sedimentary C stocks. However, these interpretations should be taken with caution as sample size within locations and habitats was relatively low.

The results here suggest that the various degrees of development across the three estuaries in this study had no significant influence on C stocks per unit area among each blue carbon system. There was however, a significant difference among replicate C stock estimates within each blue carbon system in each estuary; suggesting that local conditions may influence the variabilities more so than regional scale processes. Local scale conditions such as elevation and subsequent tidal inundation, geomorphic location within the estuary, and proximity to allochthonous carbon sources as well as nutrient inputs have been identified in the literature to be major drivers of sedimentary blue carbon stock variabilities (McLeod et al., 2011; Sanders et al., 2014). Although the causes of variability in sedimentary C stocks estimates in this study are not clear, a significantly negative relationship was noted between the carbon content and the DBD (**Figure 3**). This relation suggests that DBD of blue carbon sediments are directly related to the organic matter as also described by Morris et al. (2016) for tidal wetlands.

The sedimentary C stock data from this study adds to the growing understanding of regional C storage in coastal blue carbon systems. While we have shown C stocks per area in seagrass, mangroves and saltmarsh sediments to be similar in subtropical estuaries with various stages of development (**Figure 2**), the effects of development itself on the areal extent of these systems can have severe consequences in regards to the

TABLE 4 | Total C stock values from each blue C system in this study and associated monetary values as well as potential CO_2 emissions.

Study site	Blue carbon system	Total C stock (Mg C)	Monetary value ($AUD)	Approximate emissions from top 30 cm of sediment (Mg CO_2 eq)			Approximate emissions from top 100 cm of sediment (Mg CO_2 eq)		
				43% loss	50% loss	90% loss	43% loss	50% loss to 30 cm then 25% loss to 1 m	90% loss
Wooli River estuary	Seagrass	3,489 ± 291	42,359 ± 3,536	551 ± 46	640 ± 53	1152 ± 96	1,835 ± 153	648 ± 54	3,842 ± 321
	Mangrove	55,267 ± 8,050	670,943 ± 97,733	10,550 ± 1,537	12,268 ± 1,787	22,082 ± 3,217	35,168 ± 5,123	12,411 ± 1,808	73,608 ± 10,722
	Saltmarsh	60,103 ± 11,531	729,646 ± 139,982	15,216 ± 2,919	17,693 ± 3,394	31,848 ± 6,110	50,721 ± 9,731	17,900 ± 3,434	106,160 ± 20,367
	Total	**118,859 ± 19,872**	**1,442,949 ± 241,251**	**26,317 ± 4,502**	**30,602 ± 5,235**	**55,083 ± 9,423**	**87,724 ± 15,007**	**30,959 ± 5,296**	**183,609 ± 31,409**
Corindi River estuary	Seagrass	1,255 ± 812	15,239 ± 9,860	290 ± 188	337 ± 218	607 ± 393	966 ± 625	341 ± 221	2,023 ± 1,309
	Mangrove	27,406 ± 3,738	332,713 ± 45,373	5,128 ± 699	5,963 ± 813	10,734 ± 1,464	17,095 ± 2,331	6,033 ± 823	35,780 ± 4,879
	Saltmarsh	23,471 ± 6,485	284,943 ± 78,730	5,528 ± 1,527	6,428 ± 1,776	11,571 ± 3,197	18,428 ± 5,092	6,503 ± 1,797	38,570 ± 10,657
	Total	**52,133 ± 11,035**	**632,896 ± 133,963**	**10,947 ± 2,414**	**12,729 ± 2,807**	**22,912 ± 5,053**	**36,489 ± 8,048**	**12,877 ± 2,840**	**76,373 ± 16,845**
Coffs Creek estuary	Seagrass	73 ± 45	886 ± 547	14 ± 9	17 ± 10	30 ± 18	48 ± 29	17 ± 10	100 ± 62
	Mangrove	21,503 ± 17,240	261,049 ± 209,293	3,393 ± 2,721	3,946 ± 3,164	7,103 ± 5,694	11,311 ± 9,069	3,992 ± 3,200	23,675 ± 18,981
	Saltmarsh	1,266 ± 265	15,370 ± 3,222	247 ± 52	287 ± 60	517 ± 108	823 ± 173	291 ± 61	1,723 ± 361
	Total	**22,842 ± 17,550**	**277,305 ± 213,062**	**3,655 ± 2,781**	**4,250 ± 3,234**	**7,649 ± 5,821**	**12,182 ± 9,271**	**4,299 ± 3,272**	**25,498 ± 19,404**
Total	Seagrass	4,818 ± 1,148	58,485 ± 13,942	855 ± 242	994 ± 282	1,789 ± 507	2,849 ± 808	1,006 ± 285	5,964 ± 1,691
	Mangrove	104,177 ± 29,028	1,264,705 ± 352,400	19,072 ± 4,957	22,177 ± 5,764	39,919 ± 10,375	63,574 ± 16,523	22,436 ± 5,831	133,063 ± 34,583
	Saltmarsh	84,840 ± 18,281	1,029,960 ± 221,934	20.992 ± 4,499	24,409 ± 5,231	43,936 ± 9,415	69,972 ± 14,995	24,694 ± 5,292	146,453 ± 31,385
	Total	**193,834 ± 48,458**	**2,353,150 ± 588,276**	**40,919 ± 47,580**	**47,580 ± 11,276**	**85,644 ± 20,298**	**136,396 ± 32,326**	**48,135 ± 11,408**	**285,480 ± 67,658**

The bold values are averages.

total C stored within an estuary. For instance, the areal extent of each blue carbon system identified in **Table 1** decreased with increasing development across the three estuaries (i.e., from the relatively pristine Wooli River estuary through to the sparsely developed Corindi River estuary and severely urbanized Coffs Creek estuary). Subsequently, the decreased areal extent of each blue carbon system resulted in the observed decrease in total C stocks with increasing development across the three estuaries (**Figure 4**).

Another important influence on coastal habitat area between the three estuaries may be related to water quality. Greater urban development has led to elevated suspended sediment and nutrient concentrations in Coffs Creek (Ryder et al., 2011, 2012). In a 2011 study of water quality, turbidity measurements at estuarine sites within Wooli and Corindi Creek were below ANZECC trigger values, while Coffs Creek estuarine sites exceeded turbidity trigger values between 20 and 33% of the sampling events. Increased turbidity may be related to inhibited light penetration to the seagrass systems along Coffs Creek, causing degradation of these seagrass habitats. In addition, nutrient concentrations of estuarine surface waters are higher in Coffs Creek than in Wooli and to a lesser extent Corindi (Ryder et al., 2011, 2012). Increases in catchment nutrient discharge promotes algal phase shift, influencing the seagrass habitat health and leading to meadow scale mortality (Orth et al., 2006; Burkholder et al., 2007; Ryder et al., 2011).

Potential Blue Carbon Emissions From Regional Development

Based on the average sedimentary carbon stocks for each blue carbon ecosystems in this study, and combined with the monetary value of carbon at a price of \$AUD 12.14 per Mg, established by the Australian Emissions Reduction Fund auctions (ERF), it is estimated that over \$AUD 2.35 ± 0.59 million worth of carbon could be stored within the top 3 m of sediments of the three small estuaries in this study alone (**Table 4**). Despite the similar C stocks per unit area between the three studied estuaries, the blue C systems of the relatively pristine Wooli River estuary are responsible for 61.3% of this monetary value, followed by the agriculturally developed Corindi River estuary (26.9%) and urbanized Coffs Creek estuary (11.8%). The greater areal extent of all three blue C systems in the Wooli River and to a lesser extent the Corindi River estuary allow for higher total C stocks as opposed to the urbanized Coffs Creek Estuary in which blue carbon system expansion or migration is limited along the estuary.

Quantifying the emissions potential (often measured in carbon dioxide equivalent, or CO_2e) of potentially remineralized C as a result of regional development and land use change is an important step toward protecting these systems and understanding these anthropogenic impacts on greenhouse gas forcing (Atwood et al., 2017). Despite this, there are uncertainties within the literature in regards to the amount of organic C that may be remineralized to CO_2 from blue C habitat degradation. For instance, Murray et al. (2011) estimated that 90% of organic C in the top meter of sediment is remineralized to CO_2 emissions while Donato et al. (2011) estimates that only 50% of organic C is remineralized from the top 30 cm and 25% of organic C is remineralized from 30 cm to 1 m depth. Furthermore, a recent study by Atwood et al. (2017) compiled carbon emissions data from several studies of blue C habitat disturbances to estimate 43% organic C remineralization to CO_2 in the top 1 m of sediment within 1 year after disturbance. For consistency and comparability, CO_2 equivalents in this study were calculated using organic C remineralization estimates from previous studies (Donato et al., 2011; Murray et al., 2011; Atwood et al., 2017) along with a conversion factor of 3.67 to account for differences in molecular weight of CO_2 compared to C (Murray et al., 2011; Lewis et al., 2017) for this study are shown on **Table 4**.

Lewis et al. (2017) estimated low and high potentials for organic C remineralization from blue C systems of Victoria, Australia to be between 2,681,271 ± 198,795 and 4,826,288 ± 357,831 Mg CO_2e (assuming 50 and 90% remineralization, respectively, to a depth of 30 cm). As indicated on **Table 4**, assuming 50 and 90% remineralization to a depth of 30 cm, the values found in this study by extrapolating New South Wales is estimated to be 1,099,081 ± 168,057 and 1, 978,347 ± 302,504 Mg CO_2e, respectively. Atwood et al. (2017) compiled carbon emissions data from several mangrove habitat disturbances to estimate 43% organic C remineralization to CO_2 in the top 1 m of sediment within 1 year after disturbance. Using this more recent emissions estimate of 43% remineralization (Atwood et al., 2017) and a conversion factor of 3.67 to CO_2e (Murray et al., 2011), we estimate that three estuaries have potential emissions of 10,458 ± 2,965, 233,318 ± 60,639, and 256,798 ± 55,032 tons CO_2e from seagrass, mangrove, and saltmarsh, respectively. It should be noted that this estimate only assumes emissions from 1 m of sediment, and during the first year after disturbance. More research should be conducted to decrease uncertainties of CO_2 emissions and the long term (>1 year after disturbance) effects of blue C habitat degradation. However, these emission estimates are useful when informing managers involved in C trading schemes, such as the ERF.

CONCLUSION

Sediment cores from seagrass, mangrove, and saltmarsh habitats revealed high variability and similarity in organic carbon stocks of a nearly pristine Wooli River estuary as compared to the nearby agriculturally developed Corindi River estuary and urbanized Coffs Creek estuary. These results indicate similar C stocks per area between the three locations within the same region (within 50 km). Furthermore, even though the total carbon stocks decrease with increasing development, information in terms of loss of vegetated area as a result of land use change is not available, which would be important to consider for comparative purposes between each estuary studied and the effects of development. Based on the assumptions outlined in this manuscript, the relatively small regional estuaries have potential emissions of

500,574 ± 118,635 tons of CO_2e. This study highlights that the presence of more vegetation due to less development provides a greater area for C storage by incremental increases in total estuary sedimentary blue carbon stocks. However, urban blue C habitats (such as Coffs Creek) and agriculturally impacted blue C habitats (such as Corindi) also have the potential to continue providing valuable ecosystem services. Therefore, the conservation and restoration of blue C habitats in both urban and less developed estuaries are justified to maximize organic C sequestration and reduce atmospheric C emissions.

AUTHOR CONTRIBUTIONS

All authors did the field work and wrote the manuscript.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2018.00518/full#supplementary-material

REFERENCES

ABS (2016). *Census of Population and Housing*. Canberra, ACT: Australian Bureau of Statistics.

Alongi, D. M. (2002). Present state and future of the world's mangrove forests. *Environ. Conserv.* 29, 331–349. doi: 10.1017/S0376892902000231

Anderson, M. J. (2008). Animal-sediment relationships re-visited: characterising species' distributions along an environmental gradient using canonical analysis and quantile regression splines. *J. Exp. Mar. Biol. Ecol.* 366, 16–27, doi: 10.1016/j.jembe.2008.07.006

Atwood, T. B., Connolly, R. M., Almahasheer, H., Carnell, P. E., Duarte, C. M., Lewis, C. J. E., et al. (2017). Global patterns in mangrove soil carbon stocks and losses. *Nat. Clim. Chang.* 7, 523–528. doi: 10.1038/nclimate3326

Beaumont, N. J., Jones, L., Garbutt, A., Hansom, J. D., and Toberman, M. (2014). The value of carbon sequestration and storage in coastal habitats. *Estuar. Coast. Shelf Sci.* 137, 32–40. doi: 10.1016/j.ecss.2013.11.022

Brown, D. R., Conrad, S., Akkerman, K., Fairfax, S., Fredericks, J., Hanrio, E., et al. (2016). Seagrass, mangrove and saltmarsh sedimentary carbon stocks in an urban estuary; Coffs Harbour, Australia. *Reg. Stud. Mar. Sci.* 1, 1–6. doi: 10.1016/j.rsma.2016.08.005

Burkholder, J. M., Tomasko, D. A., and Touchette, B. W. (2007). Seagrasses and eutrophication. *J. Exp. Mar. Biol. Ecol.* 350, 46–72. doi: 10.1016/j.jembe.2007.06.024

Chmura, G. L., Anisfeld, S. C., Cahoon, D. R., and Lynch, J. C. (2003). Global carbon sequestration in tidal, saline wetland soils. *Glob. Biogeochem. Cycles* 17:1111. doi: 10.13227/j.hjkx.2016.06.049

Clarke, K. R., and Gorley, R. N. (2015). *Getting Started with PRIMER v7. PRIMER-EPlymouth*.

Conrad, S. R., Sanders, C. J., Santos, I. R., and White, S. A. (2018). Investigating water quality in Coffs coastal estuaries and the relationship to adjacent land use. Part 1: Sediments. National Marine Science Centre, Southern Cross University, Coffs Harbour, NSW, 42.

Duarte, C. M., Losada, I. J., Hendriks, I. E., Mazarrasa, I. and Marbà, N. (2013). The role of coastal plant communities for climate change mitigation and adaptation. *Nat. Clim. Chang.* 3, 961–968. doi: 10.1038/nclimate1970

Donato, D. C., Kauffman, J. B., Murdiyarso, D., Kurnianto, S., Stidham, M., and Kanninen, M. (2011). Mangroves among the most carbon-rich forests in the tropics. *Nat. Geosci.* 4, 293–297. doi: 10.1038/ngeo1123

Friess, D. A., and Webb, E. L. (2014). Variability in mangrove change estimates and implications for the assessment of ecosystem service provision. *Glob. Ecol. Biogeogr.* 23, 715–725. doi: 10.1111/geb.12140

Gedan, K. B., Silliman, B., and Bertness, M. (2009). Centuries of human-driven change in salt marsh ecosystems. *Mar. Sci.* 1, 117–141. doi: 10.1146/annurev.marine.010908.163930

Hayes, M. A., Jesse, A., Hawke, B., Baldock, J., Tabet, B., Lockington, D., et al. (2017). Dynamics of sediment carbon stocks across intertidal wetland habitats of Moreton Bay, Australia. *Glob. Chang. Biol.* 23, 4222–4234. doi: 10.1111/gcb.13722

Hopkinson, C. S., Cai, W.-J., and Hu, X. (2012). Carbon sequestration in wetland dominated coastal systems — a global sink of rapidly diminishing magnitude. *Curr. Opin. Environ. Sustain.* 4, 186–194. doi: 10.1016/j.cosust.2012.03.005

Howard, J., Mcleod, E., Thomas, S., Eastwood, E., Fox, M., Wenzel, L., et al. (2017). The potential to integrate blue carbon into MPA design and management. *Aquat. Conserv.* 27, 100–115. doi: 10.1002/aqc.2809

Irving, A. D., Connell, S. D., and Russell, B. D. (2011). Restoring coastal plants to improve global carbon storage: reaping what we sow. *PLoS One* 6:e18311. doi: 10.1371/journal.pone.0018311

Kelleway, J., Serrano, O., Baldock, J., Cannard, T., Lavery, P., Lovelock, C., et al. (2017). *Technical Review of Opportunities for Including Blue Carbon in the Australian Government's Emissions Reduction Fund*. Canberra, ACT: CSIRO.

Lavery, P. S., Mateo, M. -Á, Serrano, O., and Rozaimi, M. (2013). Variability in the carbon storage of seagrass habitats and its implications for global estimates of blue carbon ecosystem service. *PLoS One* 8:e73748. doi: 10.1371/journal.pone.0073748

Lewis, C. J. E., Carnell, P. E., Sanderman, J., Baldock, J. A., and Macreadie, P. I. (2017). Variability and vulnerability of coastal 'blue carbon'stocks: a case study from Southeast Australia. *Ecosystems* 21, 263–279. doi: 10.1007/s10021-017-0150-z

Lovelock, C. E., Atwood, T., Baldock, J., Duarte, C. M., Hickey, S., Lavery, P. S., et al. (2017). Assessing the risk of carbon dioxide emissions from blue carbon ecosystems. *Front. Ecol. Environ.* 15:257–265. doi: 10.1002/fee.1491

Lovelock, C. E., Cahoon, D. R., Friess, D. A., Guntenspergen, G. R., Krauss, K. W., Reef, R., et al. (2015). The vulnerability of Indo-Pacific mangrove forests to sea-level rise. *Nature* 526, 559–563. doi: 10.1038/nature15538

Macreadie, P. I., Nielsen, D. A., Kelleway, J. J., Atwood, T. B., Seymour, J. R., Petrou, K., et al. (2017a). Can we manage coastal ecosystems to sequester more blue carbon? *Front. Ecol. Environ.* 15, 206–213. doi: 10.1002/fee.1484

Macreadie, P. I., Ollivier, Q., Kelleway, J., Serrano, O., Carnell, P., Lewis, C. E., et al. (2017b). Carbon sequestration by Australian tidal marshes. *Sci. Rep.* 7:44071. doi: 10.1038/srep44071

Maher, D. T., Call, M., Santos, I. R., and Sanders, C. J. (2018). Beyond burial: lateral exchange is a significant atmospheric carbon sink in mangrove forests. *Biol. Lett.* 14:20180200. doi: 10.1098/rsbl.2018.0200

McLeod, E., Chmura, G. L., Bouillon, S., Salm, R., Björk, M., Duarte, C. M., et al. (2011). A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO 2. *Front. Ecol. Environ.* 9, 552–560. doi: 10.1890/110004

Morris, J. T., Barber, D. C., Callaway, J. C., Chambers, R., Hagen, S. C., Hopkinson, C. S., et al. (2016). Contributions of organic and inorganic matter to sediment volume and accretion in tidal wetlands at steady state. *Earth's Future* 4, 110–121. doi: 10.1002/2015EF000334

Murray, B. C., Pendleton, L., Jenkins, W. A., and Sifleet, S. (2011). *Green Payments for Blue carbon: Economic Incentives for Protecting Threatened Coastal Habitats*. Durham: Nicholas Institute for Environmental Policy Solutions.

Orth, R. J., Carruthers, T. J., Dennison, W. C., Duarte, C. M., Fourqurean, J. W., Heck, K. L., et al. (2006). A global crisis for seagrass ecosystems. *AIBS Bull.* 56, 987–996. doi: 10.1641/0006-3568(2006)56[987:AGCFSE]2.0.CO;2

Pendleton, L., Donato, D. C., Murray, B. C., Crooks, S., Jenkins, W. A., Sifleet, S., et al. (2012). Estimating global "blue carbon" emissions from conversion and degradation of vegetated coastal ecosystems. *PLoS One* 7:e43542. doi: 10.1371/journal.pone.0043542

Ricart, A. M., York, P. H., Rasheed, M. A., Pérez, M., Romero, J., Bryant, C. V., et al. (2015). Variability of sedimentary organic carbon in patchy seagrass landscapes. *Mar. Pollut. Bull.* 100, 476–482. doi: 10.1016/j.marpolbul.2015.09.032

Roper, T., Creese, B., Scanes, P., Stephens, K., Williams, R., Dela-Cruz, J., et al. (2011). "Assessing the condition of estuaries and coastal lake ecosystems," in *NSW Monitoring, Evaluation and Reporting Program*, (Sydney: Office of Environment and Heritage).

Ryder, D., Burns, A., Veal, R., Schmidt, J., Robertson, M., Stewart, M., et al. (2012). *Coffs Creek Estuary Coastal Zone Management Plan Literature and Information Review*, ed. O. O. E. A. Heritage (Coffs Harbour, NSW: Coffs Harbour City Council).

Ryder, D., Veal, R., Sbrocchi, C., and Schmidt, J. (2011). *Bellinger-Kalang Rivers EcohealthProject: Assessment of River and Estuarine Condition 2009-2010*. Bellingen, NSW: Bellingen Shire Council.

Sanders, C. J., Eyre, B. D., Santos, I. R., Machado, W., Luiz-Silva, W., Smoak, J. M., et al. (2014). Elevated rates of organic carbon, nitrogen, and phosphorus accumulation in a highly impacted mangrove wetland. *Geophys. Res. Lett.* 41, 2475–2480. doi: 10.1002/2014GL059789

Sanders, C. J., Maher, D. T., Tait, D. R., Williams, D., Holloway, C., Sippo, J. Z., et al. (2016). Are global mangrove carbon stocks driven by rainfall? *J. Geophys. Res.* 121, 2600–2609. doi: 10.1002/2016JG003510

Sanders, C. J., Smoak, J. M., Waters, M. N., Sanders, L. M., Brandini, N., and Patchineelam, S. R. (2012). Organic matter content and particle size modifications in mangrove sediments as responses to sea level rise. *Mar. Environ. Res.* 77, 150–155. doi: 10.1016/j.marenvres.2012.02.004

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