

# Allochthonous and autochthonous contributions to carbon accumulation and carbon store in southeastern Australian coastal wetlands



N. Saintilan<sup>a,\*</sup>, K. Rogers<sup>b</sup>, D. Mazumder<sup>c</sup>, C. Woodroffe<sup>b</sup>

<sup>a</sup> Science Division, NSW Office of Environment and Heritage, PO Box A290, Sydney South, NSW 1232, Australia

<sup>b</sup> School of Earth and Environmental Science, University of Wollongong, Australia

<sup>c</sup> Australian Nuclear Science and Technology Organisation (ANSTO), Australia

## ARTICLE INFO

### Article history:

Received 15 February 2013

Accepted 11 May 2013

Available online 21 May 2013

### Keywords:

carbon sequestration

mangrove

saltmarsh

stable isotope

geomorphology

in-situ

tidal

## ABSTRACT

Estimates of carbon store and carbon accumulation rate in mangrove and saltmarsh are beset by issues of scale and provenance. Estimates at a site do not allow scaling to regional estimates if the drivers of variability are not known. Also, carbon accumulation within soils provides a net offset only if carbon is derived in-situ, or would not otherwise be sequestered. We use a network of observation sites extending across 2000 km of southeastern Australian coastline to determine the influence of geomorphic setting and coastal wetland vegetation type on rates of carbon accumulation, carbon store and probable sources. Carbon accumulation above feldspar marker horizons over a 10-year period was driven primarily by tidal range and position in the tidal frame, and was higher for mangrove and saltmarsh dominated by *Juncus kraussii* than for other saltmarsh communities. The rate of carbon loss with depth varied between geomorphic settings and was the primary determinant of carbon store. A down-core enrichment in  $\delta^{13}\text{C}$  was consistent with an increased relative contribution of mangrove root material to soil carbon, as mangrove roots were found to be consistently enriched compared to leaves. We conclude that while surface carbon accumulation is driven primarily by tidal transport of allochthonous sediment, in-situ carbon sequestration is the dominant source of recalcitrant carbon, and that mangrove and saltmarsh carbon accumulation and store is high in temperate settings, particularly in mesotidal and fluvial geomorphic settings.

© 2013 Elsevier Ltd. All rights reserved.

## 1. Introduction

Mangrove and saltmarsh are among the most efficient ecosystems in the world at sequestering carbon (Pidgeon, 2009; Donato et al., 2011), a capacity that places these wetlands at the forefront of scientific and policy interest. This is because methane emissions are dramatically reduced in environments where methanogenic bacteria are inhibited by salt (Poffenbarger et al., 2011), and because the biogeochemical conditions in tidal wetlands are conducive to long-term carbon retention (DeLaune et al., 1990; Choi and Wang, 2004). Reviews by Duarte et al. (2005) and Bouillon et al. (2008) report a mean organic carbon burial rate of  $1.51 \text{ Mg ha y}^{-1}$  for saltmarsh (maximum  $17.2 \text{ Mg ha y}^{-1}$ ) and  $1.39 \text{ Mg ha y}^{-1}$  for mangrove (maximum  $6.54 \text{ Mg ha y}^{-1}$ ), exceeding by 10 and 6 times

respectively the maximum burial rate of undisturbed Amazonian forest ( $1.02 \text{ Mg ha y}^{-1}$ ) (Grace et al., 1993; Nellemann et al., 2009).

Considerable progress has been made in developing standard methodologies for carbon benefits associated with terrestrial forests. Programs and incentives under the REDD + framework (Reduced Emission from Deforestation and Degradation) provide a mechanism for compensating countries for the avoidance of carbon losses to the atmosphere resulting from deforestation and degradation (Anglesen, 2009). However, REDD + has not to date been extended to coastal wetlands, even though the carbon store of coastal mangrove and saltmarshes are potentially large, and their rate of decline in extent at a global scale has been rapid in modern times. Improved estimates of coastal mangrove and saltmarsh carbon stores and fluxes are required to meet obligations to monitor, report and verify sequestration needed for carbon financing programs such as REDD+ and emission trading schemes (Alongi, 2011).

The inclusion of coastal wetlands in carbon accounting, trading and incentive schemes will be contingent upon improved estimates of the carbon store and rates of flux in coastal wetlands, and an

\* Corresponding author.

E-mail address: [neil.saintilan@environment.nsw.gov.au](mailto:neil.saintilan@environment.nsw.gov.au) (N. Saintilan).

improved understanding of drivers of variability in carbon sequestration potential. There are *a priori* reasons for supposing mangrove and saltmarsh below-ground biomass and carbon stores vary between geomorphic and geographic settings, given differences in productivity forced by temperature and nutrient levels (Saenger et al., 1993). For example, Chmura et al. (2003) compiled data on mangrove and saltmarsh carbon accumulation rates for 154 sites, predominantly in North America, but including some Oceanic, Indian, and European locations. They found evidence that soil carbon density declines with increasing air temperature for *Spartina*, but otherwise variation in sequestration rate was associated with variability in sedimentation rates.

Understanding factors relating to variation in sequestration rates at regional scales will be important for national carbon accounting, as well as targeting the application of incentive and wetland restoration schemes. Coastal wetlands of southeast Australia provide a natural testing ground to investigate drivers of variability as they occupy a range of geomorphic types and span subtropical to cool temperate climates. Mangrove and saltmarsh co-exist across the entire region, with mangroves generally occupying the lower, more frequently inundated position within the tidal prism. Prior studies in the region indicate that the estuaries of Westernport Bay (Livesley and Andrusiak, 2012) and Hunter River (Howe et al., 2009) both show higher carbon store in saltmarsh than mangrove. Howe et al. (2009) used feldspar marker horizons to estimate the rate of carbon accumulation in the Hunter intertidal wetlands, with estimates low by global standards, though higher in a restoration site compared to an undisturbed site.

Howe et al. (2009) proposed a carbon store for coastal wetlands of 3900–5600 GgC for New South Wales. However, there are three important caveats in accepting this estimate. First, the estimate of soil carbon store was based on sampling of the upper 20 cm of soil, which may be insufficient to account for losses of carbon down profile encountered in other studies (Wang et al., 2011; Livesley and Andrusiak, 2012). Second, it is unclear whether the accumulation rates derived from feldspar horizons, as described by Howe et al. (2009) and Chmura et al. (2003), consist of allochthonous or autochthonous carbon, a potentially important consideration in the context of carbon emission offsets. Third, the estimate does not incorporate potential variability between geomorphic and habitat settings encountered across the region.

In this study we assess carbon sequestration for mangrove and saltmarsh where they co-exist across 2000 km of coastline in southeastern Australia, incorporating measures from a greater range of vegetation communities and geomorphic settings than any previously published account. Geomorphic setting integrates a range of geochemical conditions relevant to growth of vegetation and carbon accumulation and efflux, as well as geomorphic history as an influence on down-core changes in carbon density and source, while vegetation community composition is an important potential driver of carbon accumulation. Our aim is to determine the relative importance of vegetation communities, frequency of inundation, tidal range and temperature in forecasting carbon store in the upper metre of sediment and surficial carbon sequestration rates for mangrove and saltmarsh at regional scales.

We also seek to identify, through field studies and isotopic analysis, the likely sources of carbon being accumulated, specifically, whether it is autochthonous or allochthonous, as this is an important consideration in carbon sequestration benefit of restored or protected wetlands. Stable isotopes have been used to clarify the sources of carbon, utilising differences in isotopic composition of C<sub>3</sub> and C<sub>4</sub> plants, and aquatic submerged plants (seagrasses) that derive carbon from the water column. Studies in forest and coastal wetland systems have noted down-core enrichment in the heavier isotope (Garten et al., 2007). In terrestrial studies this has been

associated with the loss of carbon resulting from microbial activity and decomposition, and hence providing a potential indicator of the rate of decay of organic carbon and the depth at which carbon store is stable (Ehleringer et al., 2000; Garten, 2006). However, previous studies in coastal wetlands have emphasised the importance of vegetation change in interpreting down-core  $\delta^{13}\text{C}$  (Choi et al., 2001; Wang et al., 2011) with  $\delta^{13}\text{C}$  enrichment associated with communities dominated by C<sub>4</sub> photosynthesising plants.

Our study samples carbon accumulation and below-ground carbon store across a range of geomorphic settings and vegetated habitats in southeastern Australia. We test the following hypotheses:

- the rate of surface carbon accumulation is related to vegetation type and geomorphic setting; specifically that where mangrove and saltmarsh will show consistent differences in the rate of carbon accumulation. We use a 10-year record of surface accumulation across 10 sites to test this hypothesis.
- carbon store, as defined by carbon density in the upper metre of sediment, varies predictably between habitat types and geomorphic settings, with higher store in mangrove than saltmarsh.
- carbon density decreases down-core, reflecting the loss of carbon from the profile over time.
- changes in carbon isotope ratios within the core are consistent with carbon derived primarily from local plant material.

## 2. Methods

### 2.1. Study sites

The estuaries of southeastern Australia, in particular, have been described by Roy (1984), who recognised three types: drowned river valleys; barrier estuaries; and saline coastal lakes, which have more recently been termed: tide-dominated; wave-dominated; and intermittently closed (Roy et al., 2001). The classification, and subsequent conceptual models of ecological functioning and geomorphic development of these systems, was based on a detailed program of coring and radiometric dating which enabled the identification of successive evolutionary stages.

The larger wave-dominated estuaries can be broadly divided into three zones (Dalrymple et al., 1992). The size and shape of the features in each of the three zones is controlled by the form of the paleo-valley and the degree to which the estuary has infilled. The head of the estuary is characterised by a fluvial, or bay-head, delta in which the river (or creek) channel is flanked by a floodplain and may contain channel bars. The fluvial delta is built from alluvial sediment and protrudes into the estuary, developing a birds-foot, lobate or cusped morphology depending on the relative magnitude of wave and tidal processes at this landward margin of the central basin. The central regions of the estuary form a mud basin, comprising a low-energy environment where silt and clay transported from the land accumulates. This central basin fills in progressively and may be fringed by subtidal to intertidal deposits with mangroves and saltmarshes. The mouth of a barrier estuary comprises a sand barrier which separates the estuary from the open ocean. This generally contains a sandy barrier/spit, beach ridges, and an entrance channel with ebb and/or flood tidal deltas, which protrude seaward or landward of the inlet, respectively. These environments are typically comprised of sandy sediments, derived from the nearshore, because of the relatively high wave energy at the mouth and tide energy in the inlet, and contrast with the more lithic, less angular sands that are delivered from the catchment by the river.

Individual estuaries progress through successive stages of infill, from open basins, to shallower waterways with cut-off embayments and fringing coastal wetlands, to largely infilled floodplains, termed mature riverine estuaries by Roy et al. (2001). At this final stage, the river is channelised and flows through the former estuarine reaches, generally between sandy levees to deliver fluvial sands directly to the nearshore, and mangroves and related wetlands are restricted to tributaries and incompletely filled cut-off embayments (Heap et al., 2004). Estuary type and phase of infill places important controls on estuarine hydrology which in turn influences a range of soil chemical properties relevant to the distribution of wetland species and their growth, most notably soil water content, nutrient status and salinity (Saintilan, 1998; Roy et al., 2001).

The sites chosen for an assessment of rates of carbon accumulation are distributed from the southern limit of mangrove distribution in Australia (Westernport Bay, Victoria), to the Tweed River on the NSW/Queensland border (Fig. 1, Table 1). Geomorphic settings sampled included a coastal embayment (Westernport Bay) with approximately 3 m maximum tidal range; drowned river valleys (Parramatta River, Hawkesbury River) with approximately 2 m maximum tidal range; barrier estuaries with full tidal exchange (Hunter River, Tweed River) and tidal range of approximately 2 m; and barrier estuaries with restricted tidal exchange (Cararma Inlet, Currambene Creek, Minnamurra estuary), and microtidal range.

We sampled within the dominant intertidal vegetation communities at each location. For the purposes of this study we defined three vegetation communities common across settings, these being mangrove, saltmarsh rushes dominated by *Juncus kraussii*, and saltmarsh herbfields dominated by *Sporobolus virginicus* and *Sarcocornia quinqueflora*.

## 2.2. Field sampling methods: sediment accumulation carbon store and carbon source

Feldspar marker horizons were installed in 0.25 m<sup>2</sup> plots at nine replicate locations within each vegetation community surveyed at each site (46 locations, 414 plots). We installed the feldspar marker horizons in 2000–2001 and followed the rate of sediment accumulation for 10 years, by periodically extracting small shallow cores from each plot, and measuring accretion above the feldspar layer using a ruler (following techniques of Cahoon et al., 2002). As some marker horizons dispersed due to bioturbation, results were pooled to establish accretion trajectories and identify rates of vertical accretion for each vegetation community at each site.

Three larger surface cores (0.2 m) were extracted from the vicinity of the feldspar plots, and used for the measurement of carbon density in support of the surface carbon accumulation estimates. Data from Livesley and Andrusiak (2012) were used for this purpose in Westernport Bay. In addition, deep (1.0 m) core samples were collected in replicates of three from mangrove and saltmarsh at a subset of sites covering the contrasting geomorphic settings of fluvial muds and marine sands: the Hunter River; Hawkesbury River (Berowra Creek and Marramarra Creek); and Jervis Bay (Currambene Creek and Cararma Inlet) (i.e., 5 sites, 10 locations, 3 cores per location: Fig. 2). Cores were sectioned in the field at depths of 0–10 cm, 20–30 cm, 50–60 cm and 80–90 cm. Sections were capped and frozen until further analysis. In preparation for laboratory analysis of samples, core samples were carefully removed from tubes; they were then defrosted and free water was allowed to drain away over a 12 hour period. Core length and wet sample weight was recorded.

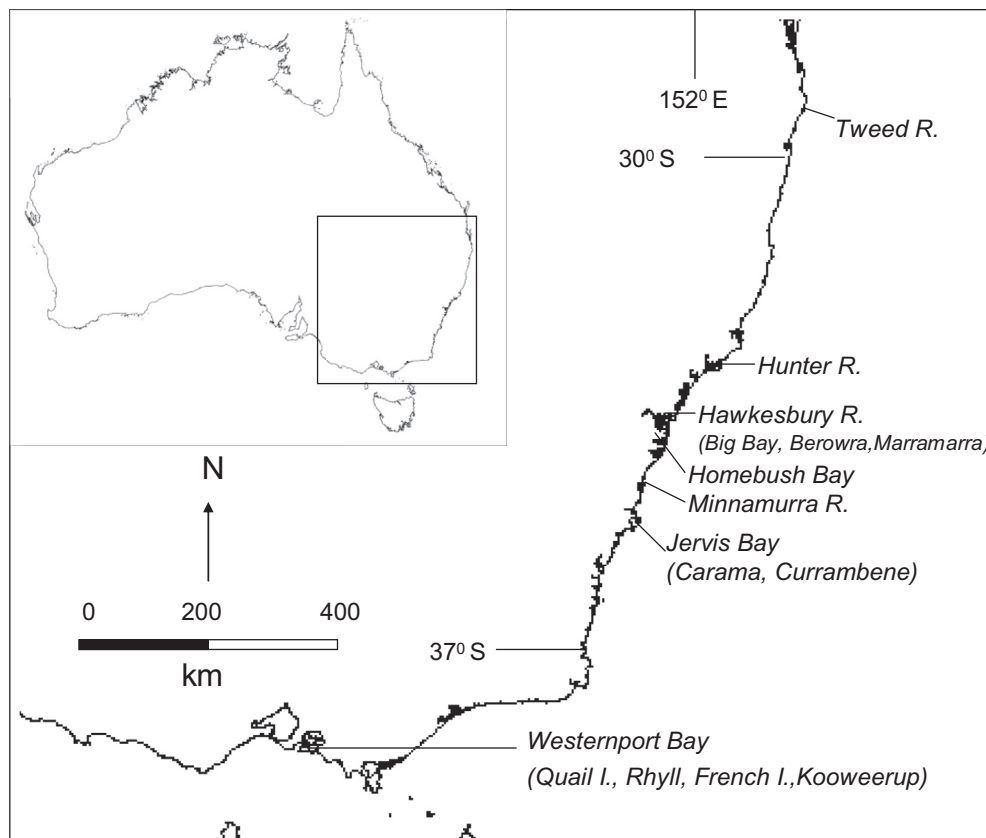


Fig. 1. Location of estuaries within which sampling sites were established.

**Table 1**  
Characteristics of study sites (See Fig. 1 for location).

Site	Geomorphic setting	Dominant mangrove	Dominant saltmarsh	Feldspar horizons	SOM sample depth
Ukerebagh Island, Tweed River	Marine Sand delta, saline	<i>Rhizophora stylosa</i> , <i>Avicennia marina</i>	<i>Sporobolus virginicus</i>	Mangrove 9 Saltmarsh 9	0.2 m
Kooragang Island (Hunter River)	Fluvial silt delta, brackish to saline	<i>Avicennia marina</i>	<i>Sporobolus virginicus</i>	Mangrove 9 Saltmarsh 9	1.0 m
Berowra Creek, Hawkesbury River	Tributary fluvial deltaic silts, brackish	<i>Avicennia marina</i>	<i>Juncus kraussii</i>	Mangrove 9 Saltmarsh 9	1.0 m
MarraMarra Ck, Hawkesbury River	Tributary fluvial deltaic silts, brackish	<i>Avicennia marina</i>	<i>Juncus kraussii</i>	Mangrove 9 Saltmarsh 9	1.0 m
Big Bay, Hawkesbury River	Mud/silt basin, brackish	<i>Avicennia marina</i>	<i>Juncus kraussii</i>	Mangrove 9 Saltmarsh 9	0.2 m
Homebush Bay, Parramatta River	Restored Tributary delta, brackish/saline	<i>Avicennia marina</i>	<i>Sarcocornia quinqueflora</i>	Mangrove 9 Saltmarsh 9	0.2 m
Minnamurra estuary	Fluvial silt delta, brackish to saline	<i>Avicennia marina</i>	<i>Sarcocornia quinqueflora</i>	Mangrove 9 Saltmarsh 9	0.2 m
Currambene Creek, Jervis Bay	Marine sand delta, saline	<i>Avicennia marina</i>	<i>Sarcocornia quinqueflora</i> <i>Juncus kraussii</i>	Mangrove 9 Saltmarsh 9	1.0 m
Carama Inlet, Jervis Bay	Marine sand delta, saline	<i>Avicennia marina</i>	<i>Sarcocornia quinqueflora</i>	Mangrove 9 Saltmarsh 9	1.0 m
Westernport Bay (4 locations)	Muddy estuarine embayment, saline	<i>Avicennia marina</i>	<i>Tecticornia</i> spp.	Mangrove 36 Saltmarsh 36	1.0 m <sup>a</sup>

<sup>a</sup> Values derived from Livesley and Andrusiak (2012).

To support interpretation of carbon isotope signatures, it was necessary to survey potential carbon sources. There are several published values of autotrophic carbon signatures for wetlands of the region (Saintilan et al., 2007; Saintilan and Mazumder, 2010), however, these did not include mangrove and saltmarsh root material, or provide an indication of variation in shoot and root  $\delta^{13}\text{C}$  along salinity gradients and across geomorphic settings. The Hawkesbury estuary was used to survey variation in plant  $\delta^{13}\text{C}$  because of the wide array of geomorphic settings of contrasting physico-chemical properties (Saintilan, 1997), ranging from muddy brackish riverine environments, to saline marine sand flats. Mangrove shoot and root material was sampled across this gradient at five sites, (Fig. 2), and *Juncus kraussii* where this occurred at one site. Saltmarsh plant shoot

material was also sampled at the estuary mouth, including the C<sub>4</sub> grass *Sporobolus virginicus*, the C<sub>3</sub> rush *J. kraussii*, and the C<sub>3</sub> succulent *Sarcocornia quinqueflora*. Several studies have indicated an enrichment in  $\delta^{13}\text{C}$  with decomposition, though evidence is equivocal (Garten et al., 2007). At Allens Creek we measured carbon isotopic composition of fresh and decomposing mangrove leaf, and fresh and ancient (1900 BP) mangrove root material (Fig. 2).

2.3. Laboratory techniques: bulk density, percent carbon and delta C

Plant samples collected for isotopic fingerprinting of potential source materials were thoroughly rinsed with Milli-Q water prior

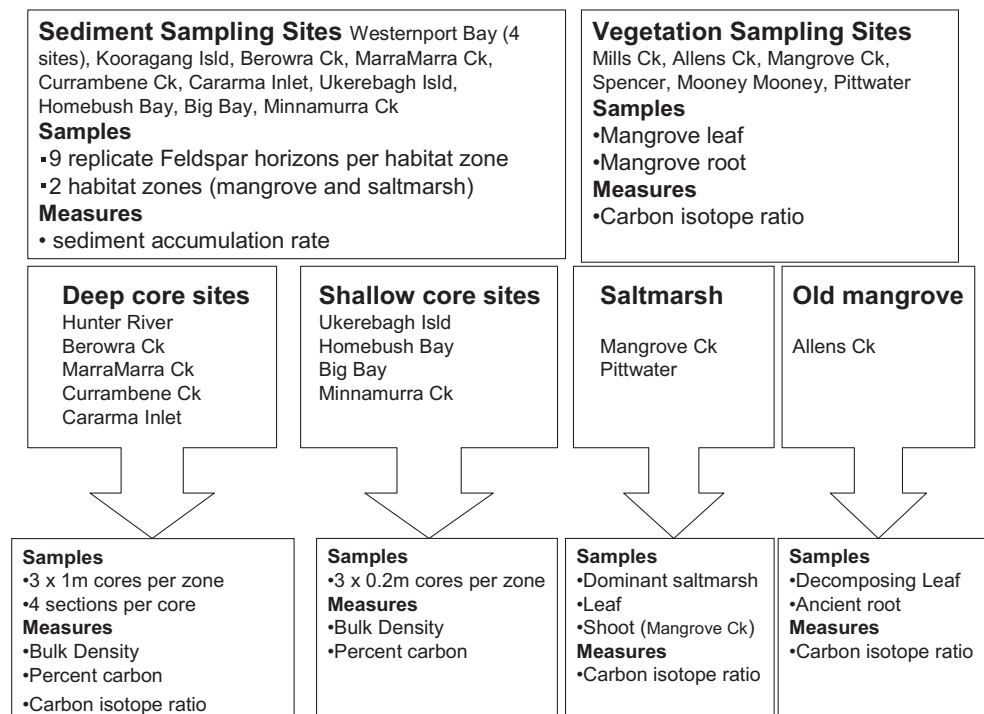


Fig. 2. Schematic of vegetation, sediment and core sampling strategy.

to processing. Leaves were cut from the plant and placed into labelled petri dishes ready for drying. Only leaf matter from each plant was used in each replica. Plant materials were oven dried at 60 °C for 72 h then ground to a fine powder with a Retsch three-dimensional Vibrator Mill (Type-MM-2:Haan, Germany) for stable isotope analysis. To determine soil bulk density, samples were oven dried to constant weight at 60 °C. Dry sample weight was determined and dry soil bulk density was estimated as the ratio between the dry sample weight (g) and the wet sample volume (cm<sup>3</sup>).

Dried sediment samples were acidified by adding 0.1 N hydrochloric acid for an hour and then gently rinsed with Milli-Q water to remove acid remaining in sediments, dried to constant weight at 60 °C and ground to a fine powder with mortar and pestle for isotopic analysis (Mazumder et al., 2010). Isotopic analysis was done at the Australian Nuclear Science and Technology Organisation (ANSTO) in Sydney, Australia. Powdered and homogenised sediment and dried plant samples were loaded into tin capsules and analysed with a continuous flow isotope ratio mass spectrometer (CF-IRMS), model Delta V Plus (Thermo Scientific Corporation, USA) interfaced with an elemental analyser (Thermo Fisher Flash 2000 HT EA, Thermo Electron Corporation, USA). The data are reported relative to IAEA secondary standards that have been certified relative to VPDB for carbon. A two point calibration is employed to normalise the data, utilising standards that bracket the samples being analysed. Two quality control references were also included in each run. Results are accurate to 1% for C % and ±0.3 per mill for δ<sup>13</sup>C. Isotopic ratios for carbon, reported as δ<sup>13</sup>C, were calculated as:

$$\delta^{13}\text{C} = \delta R_{p/\text{reference}} = \frac{R_p - R_{\text{reference}}}{R_{\text{reference}}}$$

where  $R = \%$

#### 2.4. Statistical methods

Generalised linear models provide an integrated approach to statistical analysis for data sets that may not meet the assumptions required for classical linear models, such as normal distribution (Lindsey, 1997). The technique developed by Nelder and Wedderburn (1972), enables analysis of data from a range of distributions (normal, binomial, Poisson and gamma) that are transformed to linear distribution by a link function.

Preliminary analyses were employed to identify appropriate model distributions and link functions (Table 2). Generalised linear models were employed to establish relationships between:

1. rates of vertical accretion measured derived from feldspar marker horizons, vegetation type (mangrove and saltmarsh) and tidal range at study sites
2. carbon store, vegetation community types (mangrove, *Sporobolus/Sarcocornia* saltmarsh, and *Juncus* saltmarsh), and geomorphic settings (fluvial or marine) at study sites
3. shoot and root δ<sup>13</sup>C at the five Hawkesbury sites
4. soil carbon concentration, carbon bulk density and δ<sup>13</sup>C, soil depth and vegetation type (mangrove and saltmarsh) at study sites

### 3. Results

#### 3.1. Sediment accumulation

The major driver of accretion was found to be vegetation type ( $p < 0.0001$ ), tidal range and the position of the flat in the tidal range ( $p = 0.0005$ ), though there was a significant interaction effect between tide influences and vegetation ( $p = 0.0499$ , Table 2: Model a). Sedimentation rate was higher for mangrove than saltmarsh corresponding to their differing positions in the tidal range. The rate of surface sediment accumulation was higher in *Juncus* (1.76 (±1.0) mm y<sup>-1</sup>) than in the *Sarcocornia/Sporobolus* association (1.11 (±0.08) mm y<sup>-1</sup>). No latitudinal trends were apparent in carbon density for either mangrove or saltmarsh, although the estimates for the southernmost site, Westernport Bay, were the lowest recorded in both communities.

#### 3.2. Variation in carbon and delta carbon within and between cores

The amount of carbon stored within soils (Mg ha<sup>-1</sup>) varied across three vegetation categories (mangrove > *Juncus* saltmarsh > *Sarcocornia/Sporobolus* saltmarsh;  $p < 0.0001$ ) and geomorphic setting (fluvial > marine) though the relationship was weak ( $p = 0.0699$ , Table 2: Model b). There was no carbon store interaction effect between vegetation communities and geomorphology ( $p = 0.8186$ , Table 2: Model b). Post hoc analyses indicated significant differences between *Sarcocornia/Sporobolus* saltmarsh carbon store and mangrove carbon store ( $p = 0.0027$ ), and *Juncus* saltmarsh carbon store ( $p = 0.0060$ ).

**Table 2**  
Results from generalised linear model analyses.

Model	Variables			GLM model			Significance		
	Dependent variable	Effect variable 1	Effect variable 2	Distribution	Link function	Whole model	Effect 1	Effect 2	Interaction
a	Vertical accretion	Vegetation (mangrove, saltmarsh)	Tidal range	Normal	Identity	<b><math>p &lt; 0.0001</math></b>	<b><math>p &lt; 0.0001</math></b>	<b><math>p = 0.0005</math></b>	<b><math>p = 0.0499</math></b>
b	Carbon store	Vegetation (mangrove, saltmarsh, rushes)	Geomorphic setting (fluvial, marine)	Normal	Identity	<b><math>p = 0.0002</math></b>	<b><math>p &lt; 0.0001</math></b>	$p = 0.0699$	$p = 0.8186$
c	δ <sup>13</sup> C	Soil depth (0–10 cm, 20–30 cm, 50–60 cm, 80–90 cm)	Vegetation type (mangrove, saltmarsh)	Normal	Identity	$p = 0.1379$	<b><math>p = 0.0197</math></b>	$p = 0.7594$	$p = 0.9587$
d	δ <sup>13</sup> C	Soil depth (0–10 cm, 20–30 cm, 50–60 cm, 80–90 cm)	Geomorphic setting (fluvial, marine)	Normal	Identity	<b><math>p &lt; 0.0001</math></b>	<b><math>p = 0.0014</math></b>	<b><math>p &lt; 0.0001</math></b>	<b><math>p = 0.0733</math></b>
e	% Carbon	Soil depth (0–10 cm, 20–30 cm, 50–60 cm, 80–90 cm)	Vegetation type (mangrove, saltmarsh)	Exponential	Reciprocal	<b><math>p = 0.0066</math></b>	<b><math>p = 0.0005</math></b>	$p = 0.9960$	$p = 0.8716$
f	Carbon bulk density	Soil depth (0–10 cm, 20–30 cm, 50–60 cm, 80–90 cm)	Vegetation type (mangrove, saltmarsh)	Normal	Identity	$p = 0.0625$	<b><math>p = 0.0121</math></b>	$p = 0.7852$	$p = 0.2973$
g	δ <sup>13</sup> C	Source of carbon (Avicennia root, leaf)	Study site (Allens Creek, Monemone, Pittwater, Spencer)	Normal	Identity	<b><math>p &lt; 0.0001</math></b>	<b><math>p &lt; 0.0001</math></b>	$p = 0.1518$	<b><math>p = 0.0002</math></b>

Bold values are statistically significant at  $p = 0.05$ .

Soil  $\delta^{13}\text{C}$  becomes progressively enriched with depth ( $p = 0.0197$ , Table 2: Model c). The extent of enrichment appears to interact with sites ( $p = 0.0002$ , Table 2: Model d), being highest in sandy marine settings (Fig. 3).  $\delta^{13}\text{C}$  is inversely correlated with percent carbon at all sites, with the exception of the Berowra Creek saltmarsh. Both percent carbon ( $p = 0.0005$ , Table 2: Model e) and carbon bulk density ( $p = 0.0121$ , Table 2: Model f) decreased with depth and vegetation type had little effect on this relationship. Analyses indicated that consistent down-core enrichment in  $\delta^{13}\text{C}$  (mean  $1.6 \pm 1.6\text{‰}$ ) corresponds to a loss of carbon (Table 3).

### 3.3. Carbon isotope ratios of source materials

Carbon isotope signatures divided autotrophs into two groups; relatively depleted woody plants and relatively enriched grasses. Emergent  $\text{C}_3$  woody plants occupying the mangrove and saltmarsh  $\delta^{13}\text{C}$  were depleted: *Sarcocornia quinqueflora* ( $-26.25 \pm 0.6\text{‰}$ ); *Juncus kraussii* (shoot:  $-27.03 \pm 1.8\text{‰}$ ), *Aegiceras corniculatum* ( $-26.74 \pm 1.7\text{‰}$ ) and *Avicennia marina* ( $-27.63 \pm 0.7\text{‰}$ ). The  $\text{C}_4$  grass *Sporobolus virginicus* was relatively enriched ( $-15.36 \pm 1.4\text{‰}$ ), as was the seagrass *Zostera capricorni* ( $-12.54 \pm 0.4\text{‰}$ ), which, though  $\text{C}_3$ , derived its carbon from the water column.

The grey mangrove *Avicennia marina* is the dominant plant in the region by above-ground biomass, and average leaf longevity is little more than a year (Duke, 1990). Litter fall from *A. marina* therefore represents an important potential carbon source. No difference was found in carbon isotopic composition between fresh ( $\delta^{13}\text{C}$ :  $-27.63 \pm 0.7\text{‰}$ ) and decomposing leaves ( $\delta^{13}\text{C}$ :  $-26.21 \pm 1.8\text{‰}$ ).

A significant difference was found in  $\delta^{13}\text{C}$  between *Avicennia marina* root and shoot samples across five sites of contrasting physico-chemical characteristics on the Hawkesbury River ( $p < 0.0001$  Table 2: Model g).  $\delta^{13}\text{C}$  in *A. marina* root and shoot samples did not vary between the five study sites ( $p = 0.1518$ ), but

there was a significant interaction found in  $\delta^{13}\text{C}$  in *A. marina* root or shoot samples and study sites ( $p = 0.0002$ ); this was caused by the greater difference between root and shoot  $\delta^{13}\text{C}$  in the downstream Pittwater site (Table 2: Model g). No difference was found in carbon isotopic composition between fresh roots ( $\delta^{13}\text{C}$ :  $-24.4 \pm 0.157\text{‰}$ ) and roots dating to 1700 BP ( $\delta^{13}\text{C}$ :  $-24.6\text{‰}$ ) at the Allens Creek site. No difference was detected between shoot and root  $\delta^{13}\text{C}$  for *Juncus kraussii*, the only saltmarsh species in the region forming extensive root systems (shoot  $\delta^{13}\text{C}$ :  $-27.03 \pm 1.8\text{‰}$ ; root  $-26.05 \pm 0.4\text{‰}$ ; Table 4).

## 4. Discussion

### 4.1. Variation in surface carbon accumulation rates between settings

Surface carbon accumulation rate is a function of sedimentation rate and the density of carbon within the sediment. Mangrove and *Juncus* saltmarsh showed a similar rate of carbon accumulation above feldspar horizons over a 10-year period and rates comparable to estimates globally (Table 5). Rates of carbon accumulation within the *Sporobolus/Sarcocornia* saltmarsh association were lower than the other communities sampled, and in some cases appreciably lower. This may be associated with two factors. First, the position of the saltmarsh in the upper intertidal zone receives a lower frequency, depth and duration of inundation than the lower intertidal zone occupied by mangrove. This limits the vertical accretion rate, in common with observations elsewhere (Alongi, 2009; Krauss et al., 2010; Sanders et al., 2010) and for this reason the results should not be extrapolated to sites where saltmarsh grows lower in the intertidal range. The position of saltmarsh in the tidal range explains the differences between these estimates and estimates from northern hemisphere studies cited in Chmura et al. (2003). Secondly, the productivity and biomass of the vegetation is

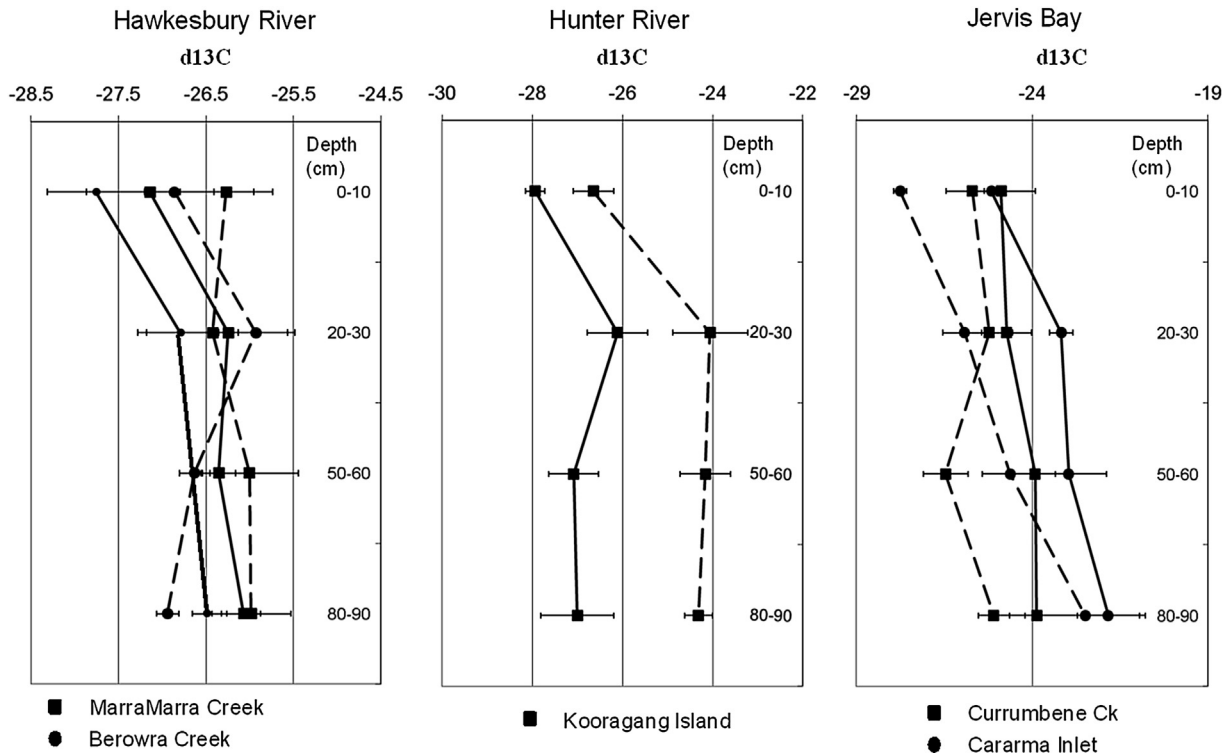


Fig. 3. Changes in  $\delta^{13}\text{C}$  with depth in sediment cores extracted from mangrove (solid line) and saltmarsh (dashed line).

**Table 3**  
Soil percentage carbon, carbon density and carbon store. Mean and standard deviation (SD).

Location	Saltmarsh					Mangrove						
	Content		C Density		$\delta^{13}\text{C}$	Content		C Density		$\delta^{13}\text{C}$		
Depth	%C	SD	g cm <sup>-3</sup>	SD	‰	SD	%C	SD	g cm <sup>-3</sup>	SD	‰	SD
<i>MarraMarra</i>												
0–10 cm	12.7	3.1	0.03	0.02	-26.3	0.5	10.0	2.0	0.031	0.002	-27.1	0.7
20–30 cm	11.3	2.5	0.032	0.004	-26.4	0.9	8.1	0.8	0.037	0.007	-26.2	0.1
50–60 cm	9.0	3.7	0.035	0.004	-26.0	0.6	6.4	3.4	0.042	0.015	-26.4	0.2
80–90 cm	8.6	2.5	0.035	0.007	-26.0	0.5	4.3	3.9	0.026	0.017	-26.1	0.2
0–100 cm	Carbon store (Mg ha <sup>-1</sup> ) 343 ± 44					Carbon store (Mg ha <sup>-1</sup> ) 343 ± 110						
<i>Berowra Creek</i>												
0–10 cm	13.9	3.8	0.026	0.004	-26.9	0.9	13.7	5.4	0.027	0.006	-27.7	0.6
20–30 cm	10.3	2.2	0.030	0.008	-25.9	0.5	10.1	1.1	0.030	0.004	-26.8	0.4
50–60 cm	12.0	1.0	0.036	0.005	-26.6	0.2	n.d.		n.d.		n.d.	
80–90 cm	8.1	0.7	0.029	0.001	-26.9	0.1	3.3	1.2	0.028	0.015	-26.5	0.2
0–100 cm	Carbon store (Mg ha <sup>-1</sup> ) 311 ± 31					Carbon store (Mg ha <sup>-1</sup> ) 285 ± 74						
<i>Kooragang Island</i>												
0–10 cm	23.5	5.4	0.031	0.001	-26.6	0.5	21.7	0.1	0.034	0.006	-27.9	0.2
20–30 cm	4.4	1.4	0.034	0.006	-24.1	0.8	6.3	2.1	0.029	0.005	-26.1	0.7
50–60 cm	0.7	0.1	0.007	0.024	-24.2	0.6	5.8	1.0	0.024	0.008	-27.1	0.6
80–90 cm	0.3	0.1	0.003	0.001	-24.3	0.3	5.1	0.9	0.024	0.001	-27.0	0.8
0–100 cm	Carbon store (Mg ha <sup>-1</sup> ) 130 ± 27					Carbon store (Mg ha <sup>-1</sup> ) 261 ± 35						
<i>Carama Inlet</i>												
0–10 cm	24.3	5.7	0.037	0.005	-27.7	0.2	22.3	6.6	0.036	0.015	-25.2	0.2
20–30 cm	0.5	0.2	0.006	0.002	-25.9	0.6	17.6	7.2	0.040	0.016	-23.2	0.3
50–60 cm	0.8	0.6	0.010	0.007	-24.6	0.8	7.5	10.5	0.018	0.012	-23.0	1.1
80–90 cm	0.6	0.2	0.008	0.001	-22.5	1.7	1.9	2.1	0.018	0.018	-21.8	0.8
0–100 cm	Carbon store (Mg ha <sup>-1</sup> ) 110 ± 26					Carbon store (Mg ha <sup>-1</sup> ) 241 ± 143						
<i>Currambene Creek</i>												
0–10 cm	13.4	7.0	0.042	0.010	-25.7	0.8	1.4	0.3	0.011	0.003	-24.9	1.0
20–30 cm	11.9	7.0	0.049	0.014	-25.2	0.7	1.9	0.6	0.013	0.003	-24.7	0.7
50–60 cm	1.8	0.6	0.013	0.002	-26.5	0.6	2.7	0.01	0.018	0.003	-23.9	0.6
80–90 cm	1.7	0.4	0.013	0.002	-25.1	0.5	2.0	0.2	0.015	0.002	-23.8	1.3
0–100 cm	Carbon store (Mg ha <sup>-1</sup> ) 61.2 ± 51					Carbon store (Mg ha <sup>-1</sup> ) 25.2 ± 20						

reportedly lower for saltmarsh than mangrove in southeastern Australia (Clarke and Jacoby, 1994), although the rate of sediment delivery may be a more important driver of sediment and carbon accumulation than *in-situ* production. The association demonstrated between tidal range and surface accumulation rate suggests most of the carbon accumulating above the feldspar horizon is allochthonous.

Our results indicate that rates of accretion are therefore dependent on both vegetation community, and the elevation of the community within the tidal range. Accretion therefore declines under stable sea-level conditions, as the elevation of the marsh increases and inundation and vegetation characteristics change (Morris et al., 2002). However, low to moderate rates of sea-level rise provide an opportunity for sustained accretion and

sequestration over periods of time meaningful for REDD + valuation. Palaeostratigraphic work by Hashimoto et al. (2006) on the Richmond River, NSW, demonstrated that mangroves were able to maintain a rate of vertical accumulation of 5 mm y<sup>-1</sup> over 900 years during the latter phases of the post-glacial marine transgression. Soil organic matter in these cores was consistently between 18 and 20%, equating to a rate of carbon sequestration of 2.75 Mg C ha<sup>-1</sup> y<sup>-1</sup>, a figure very close to the average rate estimated using contemporary data for the region (2.50 Mg C ha<sup>-1</sup> y<sup>-1</sup>). This rate of carbon sequestration is therefore sustainable over higher rates of sea-level rise than that so far encountered in southeast Australia during the historic period, though the peats were dominated by Rhizophoraceae rather than *Avicennia marina*, the dominant temperate mangrove.

**Table 4**  
Macrophyte leaf and root  $\delta^{13}\text{C}$  at 5 sites on the Hawkesbury river.

Site:	Distance from entrance (km)	Site conditions (Saintilan, 1997)	Species sampled (n = 5 unless specified)	Leaf $\delta^{13}\text{C}$ ‰ (SD)	Whole root $\delta^{13}\text{C}$ ‰ (SD)
Mills Creek	54.5	Fluvial muds and silts, brackish	<i>A. marina</i> <i>Ae. corniculatum</i>	-27.9 (1.23) -30.3 (0.063)	-24.7 (0.165)
Allens Creek	34	Fluvial muds and silts, brackish	<i>A. marina</i> Decomposing Ancient	-27.7 (0.48) -26.21 (1.8) -28.9 (0.496)	-24.4 (0.157) -24.6
Mangrove Creek	33	Estuarine silts, brackish	<i>Juncus kraussii</i> (n = 2)	-27.03 (1.8)	-26.05 (0.4)
Spencer	29.5	Estuarine silts, brackish to hypersaline-upper intertidal	<i>A. marina</i> <i>Ae. corniculatum</i>	-27.3 (0.066) -28.6 (0.16)	-24.7 (0.19)
Mooney Mooney	16.5	Estuarine silts, Saline to brackish	<i>A. marina</i>	-28.3 (0.13)	-24.3 (0.07)
Pittwater	4	Marine sand delta, sea-water salinity	<i>A. marina</i> <i>S. virginicus</i> <i>S. quinqueflora</i>	-28.3 (1.13) -15.36 (1.4) -26.25 (0.6)	-23.3 (0.34)

**Table 5**

Global estimates of carbon accumulation in mangrove and saltmarsh from this study<sup>1</sup>, Fujimoto 2004<sup>2</sup>, Chmura et al. 2003<sup>3</sup>. Australian saltmarsh includes *Juncus acutus* (j) and the *Sporobolus/Sarcocornia* association (s).

Saltmarsh		Mangrove	
Region	Mg C ha <sup>-1</sup> y <sup>-1</sup> (s.d.)	Region	Mg C ha <sup>-1</sup> y <sup>-1</sup> (s.d.)
SE Aust (j) <sup>1</sup>	2.07 (1.32)	SE Aust <sup>1</sup>	2.56 (2.24)
SE Aust (s) <sup>1</sup>	0.46 (0.37)	NE Aust <sup>3</sup>	1.48 (1.18)
NE Atlantic <sup>3</sup>	2.49 (2.02)	W Pacific <sup>3</sup>	5.4 (2.22)
Gulf of Mexico <sup>3</sup>	2.96 (3.53)	Gulf of Mexico <sup>3</sup>	2.14 (1.53)
Mediterranean <sup>3</sup>	1.61 (n = 1)	Vietnam <sup>3</sup>	5.24 (4.09)
NE Pacific <sup>3</sup>	2.06 (1.83)		
NW Atlantic <sup>3</sup>	1.74 (1.54)		

We did not measure gaseous flux in this study, however, previous studies in mangrove and saltmarsh wetlands have shown that carbon accumulation can be offset by methane emissions (Sotomayor et al., 1994; Allen et al., 2007). The only published values for southeast Australian are from Westernport Bay (Livesley and Andrusiak, 2012), where emissions from mangrove were low (between 9.7 and 405.8  $\mu\text{g C m}^{-2} \text{h}^{-1}$ ), and for saltmarsh negligible (1.8–4.2  $\mu\text{g C m}^{-2} \text{h}^{-1}$ ). Nitrous oxide emissions from both vegetation types were also low.

#### 4.2. Variation in sediment organic matter

In some circumstances, an enrichment in  $\delta^{13}\text{C}$  with depth in a coastal wetland soil profile can indicate a change in community composition from C<sub>3</sub> to C<sub>4</sub> dominated plants. Choi et al. (2001) suggested that a shift from  $-23\%$  to  $-17\%$  at depth was the result of contributions changing from *Juncus roemerianus* (C<sub>3</sub>) to C<sub>4</sub> plants at depth. The common C<sub>4</sub> saltmarsh plant on the east Australian coast *Sporobolus virginicus*, could be responsible for enrichment in cores, and this is a likely explanation of the relative enrichment of the Kooragang saltmarsh core compared to the adjacent mangrove (Fig. 3). The common seagrass plant *Zostera* is also enriched ( $\delta^{13}\text{C} \sim -12\%$ ), deriving carbon from the water column rather than atmospheric CO<sub>2</sub>. This may explain the enrichment at depth of the Currumbene Creek cores, which adjoin and may have encroached seagrass beds.

However, in most cases soil carbon is too depleted in  $\delta^{13}\text{C}$  for *Sporobolus virginicus* or seagrass to be making an important contribution. While soil carbon is enriched in comparison to mangrove and C<sub>3</sub> saltmarsh shoot material, we have demonstrated that mangrove roots are significantly enriched compared to mangrove leaves, and the profiles generally fit between the mangrove leaf and mangrove root signatures. Other studies have shown a higher rate of decomposition in leaf litter 0.28–1.53% day<sup>-1</sup> (Middleton and McKee, 2001; Poret et al., 2007) than mangrove roots (0.10–0.28% day<sup>-1</sup> (Middleton and McKee, 2001)). Down-core enrichment may be explained by the incremental loss of the relatively depleted leaf material, and the increasing contribution of recalcitrant root carbon. The difference between leaf and root  $\delta^{13}\text{C}$  is greatest in sandy marine deltaic sites, based on the Pittwater samples, where a 5‰ root enrichment was observed. This may be sufficient to explain the greater down-core enrichment of sandy marine sites at Currumbene Creek and Carama Inlet though, as indicated above, seagrass carbon may also be making a contribution.

Geomorphic setting likely exerts an influence over carbon store. Brackish fluvial environments dominated by silts and muds exhibited a greater retention of carbon down-core, especially in the two Hawkesbury River sites, and relatively high carbon density continuing to 1.0 m depth. This is likely to be a function of variation

in plant biomass between settings, and the preservation potential of the soil medium, being lower in more aerated sandy substrates. These sites maintain carbon stores of 250–350 Mg ha<sup>-1</sup>, with no difference between sites dominated by mangrove or the dominant saltmarsh in these locations, *Juncus kraussii*. Saintilan (1997) estimated *Avicennia marina* living root biomass as approximately 110 Mg C ha<sup>-1</sup> at Big Bay, Hawkesbury River, indicating a significant contribution of living material to carbon store in the upper metre. Sites situated on sandy marine deltas supported significantly lower carbon stores (25–250 Mg ha<sup>-1</sup>) regardless of vegetation type. These lower values are primarily due to the sharp loss of carbon with depth, though this does not necessarily mean carbon is lost rapidly. Carbon retrieved at 0.7 m from the Carama Inlet site dated to 1920 ± 90 BP (Saintilan and Wilton, 2001).

The results support studies applying biomarkers in affirming the importance of autochthonous carbon in saline marshes. Tanner et al. (2010) used compound-specific isotope analyses to show that saltmarsh plants dominated the carbon isotope signal of sediment organic matter in a Maine saltmarsh for more than three millenia. Our results indicate that mangrove root material possibly dominates soil organic carbon in all settings, with the exception of *Juncus* saltmarsh in brackish environments in which the relatively depleted *Juncus* roots dominate soil organic carbon.

The *Sporobolus virginicus/Sarcocornia quinqueflora* association is the most widespread saltmarsh vegetation community on the Australian east coast (Adam et al., 1988), and contributes carbon to estuarine foodwebs across the region (Guest and Connolly, 2004; Mazumder et al., 2011). The standing biomass of this community is relatively low compared to *Juncus kraussii* (Congdon and McComb, 1980; Clarke and Jacoby, 1994), and roots are shallow. Notwithstanding, the density of soil carbon within this community, though lower than the other vegetation types measured in this study, and the *Sclerostegia* saltmarsh measured by Livesley and Andrusiak (2012), is within the range of averages for saltmarshes in the northern hemisphere. Tidally-borne carbon may contribute significantly to soil organic matter in this community.

## 5. Conclusions

Our results indicate that variation in carbon store and accumulation rate on coastal wetland settings of southeastern Australia is driven primarily by hydro-geomorphic setting rather than vegetation type per se, notwithstanding the dominant contribution of in-situ vegetation to recalcitrant carbon. No difference in rate of accumulation or carbon store was observed between sites dominated by the saltmarsh *Juncus kraussii* and the mangrove *Avicennia marina*, with both communities showing strong differences between fluvial deltaic and marine deltaic settings, with the latter being appreciably lower. Accumulation rate above feldspar horizons was a function of tidal range and position in the tidal prism.

Carbon density decreased down-core, though more rapidly in sandy marine settings than fluvial mud/silt environments. The decrease in carbon was reflected in an enrichment in  $\delta^{13}\text{C}$ . Neither mangrove leaf or root material showed appreciable enrichment with decomposition. We therefore interpret down-core enrichment as an increase in the relative contribution of mangrove root material to soil carbon, given the consistent enrichment observed in mangrove root compared to mangrove leaves from the same trees across a range of settings. The result suggests an important contribution of in-situ plant material to carbon store, while surface carbon accumulation appears dominated by allochthonous sources.

Given the consistent accretion of these environments over hundreds of years (Saintilan and Hashimoto, 1999) the dominance of root carbon in soil carbon at depth suggests a relatively low residence time for allochthonous, tidally-borne carbon (including



leaf litter) in these systems. Studies of accretion using feldspar horizons should therefore be used cautiously in assessing contributions from this source to “permanent” carbon of the type required under REDD + frameworks. Several recent studies have demonstrated the potential of organic markers in clarifying sources of carbon in coastal sedimentary environments (Tanner et al., 2010; Wang et al., 2011), and the potential of compound-specific isotope analysis has not yet been fully realised in distinguishing between possible carbon sources in these settings.

## Acknowledgements

Accretion measures were funded by the Australian Commonwealth Government through the Coast and Clean Seas Initiative and the NSW Environmental Trust. Don Cahoon is thanked for providing guidance in the establishment of the measurement network. Suzanne Hollins (ANSTO) supported SI sample analysis and Barbara Neklapilova (ANSTO) and Scott Allchin (ANSTO) assisted with sample preparation and analysis.

## References

- Adam, P., Wilson, N.C., Huntley, B., 1988. The phytosociology of coastal saltmarsh vegetation in New South Wales. *Wetlands (Australia)* 7, 35–57.
- Allen, D.E., Dalal, R.C., Rennenberg, H., Meyer, R.L., Reeves, S., Schmidt, S., 2007. Spatial and temporal variation of nitrous oxide and methane flux between subtropical mangrove sediments and the atmosphere. *Soil Biology and Biochemistry* 39, 622–631.
- Alongi, D.M., 2009. The Energetics of Mangrove Forests. Springer, Dordrecht.
- Alongi, D.M., 2011. Carbon payments for mangrove conservation: ecosystem constraints and uncertainties of sequestration potential. *Environmental Science and Policy* 14, 462–470.
- Anglesen, A., 2009. Realising REDD+: National Strategy and Policy Options. Center for International Forestry Research, Bogor.
- Bouillon, S., Borges, A.V., Castañeda-Moya, E., Diele, K., Dittmar, T., Duke, N.C., Kristensen, E., Lee, S.Y., Marchand, C., Middelburg, J.J., Rivera-Monroy, V.H., Smith, T.J., Twilley, R.R., 2008. Mangrove production and carbon sinks: a revision of global budget estimates. *Global Biogeochemical Cycles* 22.
- Cahoon, D.R., Lynch, J.C., Hensel, P., Boumans, R., Perez, B.C., Segura, B., Day Jr., J.W., 2002. High-precision measurements of wetland sediment elevation: I. Recent improvements to the Sedimentation-Erosion Table. *Journal of Sedimentary Research* 72, 730–733.
- 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, 1111.
- Choi, Y., Wang, Y., 2004. Dynamics of carbon sequestration in a coastal wetland using radiocarbon measurements. *Global Biogeochemical Cycles* 18, GB4016.
- Choi, Y., Wang, Y., Hsieh, Y.-P., Robinson, L., 2001. Vegetation succession and carbon sequestration in a coastal wetland in northwest Florida: evidence from carbon isotopes. *Global Biogeochemical Cycles* 15, 311–319.
- Clarke, P.J., Jacoby, C.A., 1994. Biomass and above-ground productivity of salt-marsh plants in South-eastern Australia. *Australian Journal of Marine and Freshwater Research* 45, 1521–1528.
- Congdon, R.A., McComb, A.J., 1980. Productivity and nutrient content of *Juncus kraussii* in an estuarine marsh in south-western Australia. *Australian Journal of Ecology* 5, 221–234.
- Dalrymple, R.W., Zaitlin, B.A., Boyd, R., 1992. Estuarine facies models; conceptual basis and stratigraphic implications. *Journal of Sedimentary Research* 62, 1130–1146.
- DeLaune, R.D., Patrick, W.D., Lindau, C.W., Smith, C.J., 1990. Nitrous oxide and method emission from Gulf Coast wetlands. In: Bouwman, A.F. (Ed.), *Soils and the Greenhouse Effect*. John Wiley, New York, pp. 497–502.
- Donato, D.C., Kauffman, J.B., Murdiyoso, D., Kurnianto, S., Stidham, M., Kanninen, M., 2011. Mangroves among the most carbon-rich forests in the tropics. *Nature Geoscience* 4, 293–297.
- Duarte, C.M., Middelburg, J., Caraco, N., 2005. Major role of marine vegetation on the oceanic carbon cycle. *Biogeosciences* 2, 1–8.
- Duke, N.C., 1990. Phenological trends with latitude in the mangrove tree *Avicennia marina*. *The Journal of Ecology* 78, 113–133.
- Ehleringer, J.R., Buchmann, N., Flanagan, L.B., 2000. Carbon isotope ratios in belowground carbon cycle processes. *Ecological Applications* 10, 412–422.
- Fujimoto, K., 2004. Below-ground carbon sequestration of mangrove forests in the Asia-Pacific region. In: Vannucci, M. (Ed.), *Mangrove Management and Conservation: Present and Future*. United Nations University Press, Tokyo.
- Garten, C.T., 2006. Relationships among forest soil C isotopic composition, partitioning, and turnover times. *Canadian Journal of Forest Research* 36, 2157–2167.
- Garten, J.C., Hanson, P., Todd, J.D., Lu, B., Brice, D., 2007. Natural  $^{15}\text{N}$  and  $^{13}\text{C}$  abundance as indicators of forest N status and soil C dynamics. In: Lajtha, R.M.a.K. (Ed.), *Stable Isotopes in Ecology and Environmental Science*, second ed. Blackwell Science, Oxford.
- Grace, J., Lloyd, J., MacIntyre, J., Miranda, A.C., Meir, P., Miranda, H.S., Mobre, C., Moncrieff, J., Massheder, J., Malhi, Y., Wright, I., Gash, J., 1993. Carbon dioxide uptake by undisturbed tropical rain forest in southwest Amazonia, 1993 to 1992. *Science* 270, 778–780.
- Guest, M.A., Connolly, R.M., 2004. Fine-scale movement and assimilation of carbon in saltmarsh and mangrove habitat by resident animals. *Aquatic Ecology* 38, 599–609.
- Hashimoto, T.R., Saintilan, N., Haberle, S.G., 2006. Mid-holocene development of mangrove communities featuring Rhizophoraceae and geomorphic change in the Richmond river estuary, New South Wales, Australia. *Geographical Research* 44, 63–76.
- Heap, A.D., Bryce, S., Ryan, D.A., 2004. Facies evolution of Holocene estuaries and deltas: a large-sample statistical study from Australia. *Sedimentary Geology* 168, 1–17.
- Howe, A.J., Rodríguez, J.F., Saco, P.M., 2009. Surface evolution and carbon sequestration in disturbed and undisturbed wetland soils of the Hunter estuary, southeast Australia. *Estuarine, Coastal and Shelf Science* 84, 75–83.
- Krauss, K., Cahoon, D., Allen, J., Ewel, K., Lynch, J., Cormier, N., 2010. Surface elevation change and susceptibility of different Mangrove zones to sea-level rise on Pacific high Islands of Micronesia. *Ecosystems* 13, 129–143.
- Lindsey, J.K., 1997. *Applying Generalized Linear Models*. Springer, New York.
- Livesley, S.J., Andrusiak, S.M., 2012. Temperate mangrove and salt marsh sediments are a small methane and nitrous oxide source but important carbon store. *Estuarine, Coastal and Shelf Science* 97, 19–27.
- Mazumder, D., Iles, J., Kelleway, J., Kobayashi, T., Knowles, L., Saintilan, N., Hollins, S., 2010. Effect of acidification on elemental and isotopic compositions of sediment organic matter and macro-invertebrate muscle tissues in food web research. *Rapid Communications in Mass Spectrometry* 24, 2938–2942.
- Mazumder, D., Saintilan, N., Williams, J.R., Szymczak, R., 2011. Trophic importance of a temperate intertidal wetland to resident and itinerant taxa: evidence from multiple stable isotope analyses. *Marine and Freshwater Research* 62, 11–19.
- Middleton, B.A., McKee, K.L., 2001. Degradation of mangrove tissues and implications for peat formation in Belizean island forests. *Journal of Ecology* 89, 818–828.
- Morris, J.T., Sundareshwar, P.V., Nietch, C.T., Kjerfve, B., Cahoon, D.R., 2002. Responses of coastal wetlands to rising sea-levels. *Ecology* 83, 2869–2877.
- Nelder, J.A., Wedderburn, R.W.M., 1972. Generalized linear models. *Journal of the Royal Statistical Society* 135, 370–384.
- Nellemann, C., Corcoran, E., Duarte, C.M., Valdés, L., DeYoung, C., Fonseca, L., Grimsditch, G.E., 2009. *Blue Carbon: A Rapid Response Assessment*. United Nations Environment Programme, GRID-Arendal, Norway.
- Pidgeon, E., 2009. Carbon sequestration by coastal marine habitats: important missing sinks. In: Laffoley, D., Grimsditch, G. (Eds.), *The Management of National Coastal Carbon Sinks*. IUCN, Gland, Switzerland.
- Poffenbarger, H., Needelman, B., Megoñigal, J., 2011. Salinity influence on methane emissions from tidal marshes. *Wetlands* 31, 831–842.
- Poret, N., Twilley, R.R., Rivera-Monroy, V.H., Coronado-Molina, C., 2007. Below-ground decomposition of mangrove roots in Florida coastal Everglades. *Estuaries and Coasts* 30, 491–496.
- Roy, P.S., 1984. New south Wales estuaries: their origin and evolution. In: Thom, B.G. (Ed.), *Coastal Geomorphology in Australia*. Academic Press, Sydney, Australia, pp. 99–121.
- Roy, P.S., Williams, R.J., Jones, A.R., Yassini, I., Gibbs, P.J., Coates, B., West, R.J., Scanes, P.R., Hudson, J.P., Nichol, S., 2001. Structure and function of South-east Australian estuaries. *Estuarine, Coastal and Shelf Science* 53, 351–384.
- Saenger, P., Snedaker, S.C., 1993. Pantropical trends in mangrove above-ground biomass and annual litterfall. *Oecologia* 96, 293–299.
- Saintilan, N., 1997. Mangroves as successional stages on the Hawkesbury Rivers. *Wetlands(Australia)* 16, 99–107.
- Saintilan, N., 1998. Photogrammetric survey of the Tweed River wetlands. *Wetlands (Australia)* 17, 74–82.
- Saintilan, N., Hashimoto, R., 1999. Mangrove-saltmarsh dynamics on a prograding bayhead delta in the Hawkesbury River estuary, New South Wales, Australia. *Hydrobiologia* 413, 95–102.
- Saintilan, N., Hossain, K., Mazumder, D., 2007. Linkages between adjacent seagrass, mangrove and saltmarsh as fish habitat in the Botany Bay estuary, New South Wales. *Wetlands Ecology and Management* 15, 277–286.
- Saintilan, N., Mazumder, D., 2010. Fine-scale variability in the dietary sources of grazing invertebrates in a temperate Australian saltmarsh. *Marine and Freshwater Research* 61, 615–620.
- Saintilan, N., Wilton, K., 2001. Changes in the distribution of mangroves and saltmarshes in Jervis Bay, Australia. *Wetlands Ecology and Management* 9, 409–420.
- Sanders, C.J., Smoak, J.M., Naidu, A.S., Sanders, L.M., Patchineelam, S.R., 2010. Organic carbon burial in a mangrove forest, margin and intertidal mud flat. *Estuarine, Coastal and Shelf Science* 90, 168–172.
- Sotomayor, D., Corredor, J., Morell, J., 1994. Methane flux from mangrove sediments along the Southwestern coast of Puerto Rico. *Estuaries* 17, 140–147.
- Tanner, B.R., Uhle, M.E., Mora, C.I., Kelley, J.T., Schuneman, P.J., Lane, C.S., Allen, E.S., 2010. Comparison of bulk and compound-specific  $\delta^{13}\text{C}$  analyses and determination of carbon sources to salt marsh sediments using n-alkane distributions (Maine, USA). *Estuarine, Coastal and Shelf Science* 86, 283–291.
- Wang, J., Dodla, S., DeLaune, R., Hudnall, W., Cook, R., 2011. Soil carbon characteristics in two Mississippi river deltaic Marshland profiles. *Wetlands* 31, 157–166.