NSW SALTMARSHES ARE HOTSPOTS OF CARBON STORAGE

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Abstract

Globally, coastal saltmarshes are important sites of carbon sequestration, with an average carbon density of 162 tonnes of carbon per hectare in the surface metre of sediments (Pendleton et al. 2012). Our comprehensive study of nine estuaries along the New South Wales coastline shows carbon storage is greatest in saltmarshes dominated by fine-grained sediments and subject to fluvial inputs. Overall carbon storage is similar to global averages, suggesting NSW saltmarshes may hold more than 1.1 million tonnes of carbon. It has been estimated that NSW has lost up to 70% of its coastal wetlands since European settlement (Zann 2000). If this value is true also for saltmarshes, then projections from the current study suggest as much as 2.8 million tonnes of carbon may have been mobilised and potentially returned to the atmosphere.

Many coastal saltmarshes, however, are presently undergoing substantial change. Over the past century around one-third of NSW saltmarsh extent has been encroached or replaced by mangrove forest, due to anthropogenic and climatic changes including sea level rise. Our investigation of carbon storage shows that this saltmarsh-to-mangrove ecosystem shift leads to further increases in carbon stored aboveground (in plant biomass) and belowground (in the sediment and plant roots).

Planning policies and on-ground activities which reverse the historical decline of coastal saltmarsh and facilitate upslope migration of both saltmarsh and mangrove in response to sea level rise will increase carbon storage in coastal wetlands, presenting a negative feedback to global warming.

Keywords: blue carbon; carbon sequestration; coastal wetlands; saltmarsh; ecosystem services

Introduction

Coastal saltmarshes are intertidal ecosystems vegetated by herbs, grasses, rushes and small shrubs which provide a permanent or temporary habitat for aquatic, intertidal and terrestrial fauna. They are dynamic systems occurring in a range of sedimentary settings along low-energy coastlines and have a wide geographic distribution throughout both Australia and the world. In New South Wales (NSW), comprehensive classifications have been made of plant community structure and composition (Adam et al. 1988, Zedler et al. 1995), plant response to salinity and waterlogging (Clarke and Hannon 1970), species interactions (Clarke and Hannon 1971), relationships with mangroves (Pidgeon 1940, Mitchell and Adam 1989, Saintilan and Hashimoto 1999); and restoration procedures (Laegdsgaard 2002).

Saltmarsh communities generally have low floral species richness compared to terrestrial communities, often being dominated by one or two species (Adam et al. 1988). Along the NSW coast, for example, the majority of mid-intertidal marsh is considered a single community complex dominated by the low growing chenopod *Sarcocornia quinqueflora* and grass *Sporobulus virginicus* (Zedler et al. 1995) and upper marsh communities by the taller rush *Juncus kraussii* (Adam et al. 1988). Throughout mainland Australia, saltmarshes are generally bordered by mangroves to the lower, estuarine side and terrestrial communities to the higher, landward side.

In recent years, the ability of coastal vegetated habitats to accrete and store carbon has generated significant interest in the processes responsible for carbon sequestration in saltmarsh wetlands. Saltmarshes – along with mangroves and seagrasses - are disproportionately important in sequestering carbon relative to their spatial extent (Duarte et al. 2005, McLeod et al. 2011). Their elevated carbon storage potential is due to 1) high productivity in these ecosystems manifest in carbon bound within above and belowground biomass; 2) effective trapping of particulate carbon originating from within the ecosystem (autochthonous) or external (allochthonous) riverine and oceanic sources; and 3) anoxic, saline sub-surface conditions which slow the decay of organic material and minimise release of methane into the atmosphere (Magenheimer et al. 1996, Duarte et al. 2005, McLeod et al. 2011).

Current reviews of available data have reported a mean carbon stock of 162 ± 259 Mg C ha^{-1} in the surface metre of saltmarshes globally (Duarte et al. 2013) and an organic carbon burial rate of 151g C m^{-2} y^{-1} (Duarte et al. 2005, McLeod et al. 2011). In Australia, however, knowledge of coastal carbon stocks is limited, although advances have been made in mangrove (e.g. Alongi et al. 1998, Brunskill et al. 2002, Lovelock 2008) and seagrass (e.g. Macreadie et al. 2012) communities. Only recently has research been undertaken to investigate larger spatial variations in Australian saltmarsh and mangrove soil carbon stores. Across a single, large estuary (Hawkesbury) Saintilan et al. (2013) found soil carbon store varied between different vegetation types in the order mangrove > *Juncus* saltmarsh > *Sarcocornia/Sporobolus* saltmarsh ($p < 0.0001$); which is at odds with the previous Australian studies (i.e. Howe et al. 2009, Livesley and Andrusiak 2012). Soil carbon store also varied with geomorphic setting (fluvial > marine; though the relationship was weak $p = 0.0699$). Across all sites they concluded carbon store is high in temperate settings, particularly in mesotidal and fluvial geomorphic settings.

Similarly, recent research in Moreton Bay, Queensland, found spatial differences in saltmarsh soil carbon density with *Juncus* saltmarshes containing higher soil carbon densities than *Sarcocornia*-dominated marshes (Lovelock et al. 2013). However, attempts to ascertain carbon stock differences according to environmental setting (i.e. marine-influence oligotrophic island versus estuarine-influenced eutrophic sites) may have been confounded by differences in the vegetation composition of sites in each setting. This highlights the need for an experimental design which differentiate the relative roles of both geomorphic setting and vegetation types, as well as any interactions between the two.

The Role of Estuarine Geomorphology

In contrast to the paucity of carbon sequestration research in Australian intertidal wetlands, a number of studies have detailed the recent sedimentary history of saltmarsh and mangrove wetlands along the southeast Australian coast (Rogers et al. 2005, Rogers et al. 2006, Oliver et al. 2012) and the role of geomorphology in ecosystem form and function (Saintilan and Hashimoto 1999, Saintilan and Wilton 2001, Saintilan 2004).

Along the southeast Australian coast there are four sedimentary environments in estuaries (marine tidal delta, central mud basin, fluvial delta and riverine channel) which have characteristic water quality, nutrient cycling/primary productivity signatures and therefore ecosystems (Roy et al. 2001). Two of these geomorphic zones – marine tidal delta and fluvial delta – represent depositional environments with distinctly different sedimentary sources and salinity regimes (Figure 1). Along the central coast of NSW, saltmarsh largely occurs on deltas that develop where tributaries enter deeper waters (fluvial delta) and on back-barrier sands near the estuary mouth (marine tidal delta) (Kelleway et al. 2007, Saintilan and Rogers 2013). Consequently, these differences in geomorphic setting are likely to influence a range of geochemical conditions (e.g. salinity, nutrient and suspended particle loads) and hydrological conditions (e.g. tidal range and flow velocity) relevant to growth of vegetation as well as soil carbon accumulation.

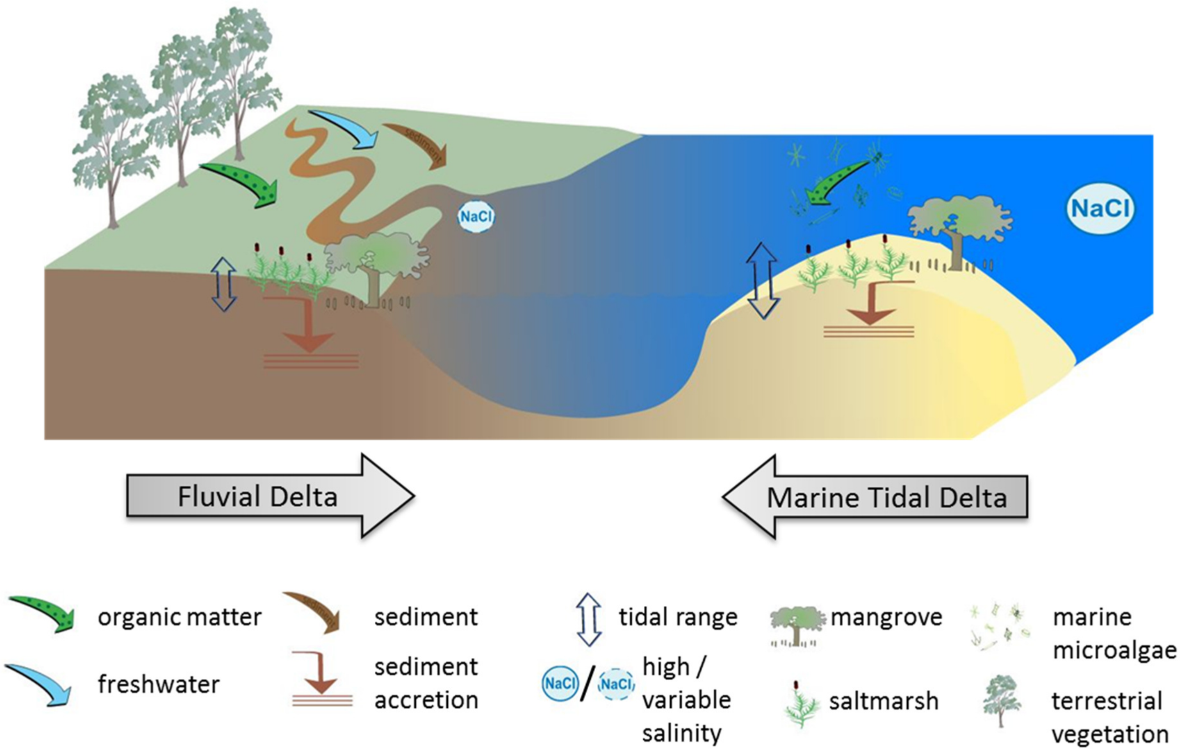

Figure 1 - Conceptual diagram of geomorphic setting influences on saltmarsh carbon dynamics

Saltmarsh / Mangrove Dynamics

Generally speaking, mangrove forests and saltmarsh meadows have opposing global distributions, whilst sharing similar ecological niches in the intertidal zone of estuaries and low-energy coastlines. Mangrove extent and species diversity is greatest in the tropics, whilst saltmarsh extent and species distribution is greater in higher latitudes. In temperate and sub-tropical zones, the two communities often overlap, forming adjacent and/or ecotonal communities.

In recent years a poleward expansion of mangrove has been reported on each of the continents mangroves and saltmarshes co-inhabit - Asia, Africa, Australia/New Zealand, North America and South America (Saintilan et al. 2014 and references therein). At the local scale, upslope mangrove incursion into saltmarsh communities is a near ubiquitous trend in south-eastern Australia, (Saintilan & Williams, 1999), and New Zealand (Saintilan et al. 2014), and has been reported in the Gulf of Mexico (Comeaux et al. 2012, Bianchi et al. 2013).

Mangroves clearly have a greater potential for carbon storage aboveground in woody components than herbaceous species (McKee and Rooth 2008). Global datasets (e.g. Chmura et al 2003) also report a higher soil carbon stock for mangrove than saltmarsh, with the most extensive research in southeast Australia following this trend. It would seem therefore, that mangrove expansion might increase carbon sequestration, as has been suggested in the Gulf of Mexico (Bianchi et al. 2013). There are however, significant questions that are yet to be answered on this topic. Among these are the relative contributions in saltmarsh and mangrove communities of below ground biomass, soil carbon source and stability, as well as the overarching role of estuarine geomorphology.

This paper

This paper summarises the outcomes of two related investigations of carbon storage in NSW intertidal ecosystems. The first is a comprehensive survey of carbon stocks along the NSW coastline, encompassing 144 sediment cores collected from two vegetation communities (rush and non-rush saltmarshes) and two geomorphic settings (marine tidal deltas and fluvial deltas). The second quantifies carbon and biomass stocks along a temporal gradient where mangroves have encroached areas previously vegetated by saltmarsh in the Botany Bay/Georges River estuary.

Methods

Saltmarsh carbon survey

Nine estuaries were selected for study on the basis of their saltmarsh aerial extent, distribution along the NSW coast, and diversity of estuary structural type (Table 1). A factorial design was implemented to understand the effects of both geomorphic setting and vegetation structure upon soil carbon stocks. Within each estuary, two sites were chosen for field sampling – one site within the marine tidal delta geomorphic zone and another site further upstream subject to fluvial influence. Sites within each of these two zones were then selected based on the presence of both of the two common saltmarsh vegetation communities of southeastern Australia: i) a rush community dominated by *J. kraussii* and/or *Baumea juncea*; ii) non-rush vegetation dominated by the grass *S. virginicus*, the succulent *S. quinqueflora* or a mosaic of the two. A sediment core of 1 m depth was collected from four locations within each vegetation zone at each site.

In the laboratory, cores were split open and sectioned for bulk density and chemical analysis. Depth intervals of 0-20, 20-50, and 50-100 cm were chosen to represent the surface rooting zone, mid and deep carbon stores, respectively. Bulk sediment was oven dried at 60°C for 72 hours and measured to determine bulk density. Sediment was homogenised and ground into a fine powder using a ball mill. Organic carbon density (g C cm^{-3}) was determined by multiplying %C by bulk density of each sediment depth interval. Total organic carbon stock (Mg C ha^{-1}) was calculated as the carbon contained within the entire 0-100 cm depth range.

Table 1. List of estuaries sampled for saltmarsh carbon stocks. Within each estuary two sites (one marine; one fluvial) were sampled with each site including rush and non-rush vegetation

Estuary	Latitude (°)	Geomorphic type	Saltmarsh area (ha)[1]
Clarence River	-29.42	Barrier estuary	290
Macleay River	-30.87	Barrier estuary	425
Lake Cathie	-31.55	Saline coastal lagoon	589
Wallis Lake	-32.17	Barrier estuary	590
Port Stephens	-32.71	Drowned valley estuary	1,063
Crookhaven River	-34.90	Barrier estuary	206
Clyde River	-35.72	Drowned valley estuary	52
Tuross Lake	-36.07	Barrier estuary	80
Wapengo Lake	-36.62	Barrier estuary	51

[1] Creese *et al.* 2009

Mangrove encroachment

A chronosequence experimental design - that is, one which uses space as a substitute for time - was developed to explore changes associated with mangrove encroachment into saltmarsh at each of two study sites in the same estuary (Towra Point Nature Reserve and Georges River National Park). For the Georges River site, aerial photographs taken in the years 1943, 1955, 1970, 1982, 1998 and 2013 were overlayed in ArcGIS. Overlaying of successive images allowed the identification of field sampling locations where mangrove encroached into saltmarsh over the following time periods relative to 2013: 0-15 y; 31-43 y; 58-70 y. For the Towra Point site, maps of vegetation change created previously using historical aerial photography (Wilton 2002) were used to identify field sampling locations corresponding to the following past time intervals: 0-14 y, 30-43 y, and 57-70 y. Interpretation of the 1943 images allowed the identification of mature mangrove stands >70 y old at each site. Locations currently vegetated exclusively by saltmarsh (*Sarcocornia quinqueflora* and *Sporobolus virginicus*) were identified in the field. Three replicate sampling locations were identified for each age zone at each site.

At each location, mangrove aboveground biomass was determined using a 100 m^2 plot. All mangrove trees >2 m height were measured for stem height (m) and diameter at breast height (DBH) (cm). Allometric equations were used to calculate aboveground biomass (kg) of all trees >2m height (Saintilan 1997) and mangrove shrubs (height <2 m) (Woodroffe 1985). Saltmarsh aboveground biomass was determined by cutting aboveground material from three 30x30 cm quadrats randomly located within 2 m of the plot centre. Material was transferred to the laboratory where it was rinsed to remove attached sediment, oven-dried at 60°C for 72 h and then weighed.

Soil cores were extracted and analysed as outlined for the statewide survey, however, a higher resolution of downcore sampling was undertaken, with samples analysed from 0-5, 5-10, 10-15, 15-20, 20-30, 30-50 and 50-100 cm.

Data Analysis

For the statewide survey, a generalised linear model was used to assess the role of geomorphic setting and vegetation structure factors on depth-integrated carbon stocks (i.e. Mg C ha^{-1}). For the mangrove encroachment study, regression analyses were performed to assess rates of change in biomass and total belowground carbon across the chronosequence.

Results and Discussion

Statewide survey

The mean belowground carbon stock of NSW saltmarshes as quantified in our study (164 Mg C ha^{-1}) is similar to the current global estimates of belowground saltmarsh stocks (162 Mg C ha^{-1}) (Duarte et al. 2013). Our range estimates is also consistent with those made previously from other estuaries in the region - (Livesley and Andrusiak 2012, Saintilan et al. 2013, Saintilan et al. 2014). Despite this similarity, however, there was substantial variability in carbon stocks observed in our regional assessment with core stocks ranging from 17.89 – 448.20 Mg C ha^{-1} with a coefficient of variation of 63%.

Depositional setting is central to saltmarsh carbon storage

Our results, show that depositional setting within an estuary – which is reflected through geomorphic setting and sediment grain size – is key to understanding variability in saltmarsh carbon stocks. First, geomorphic setting was a significant factor explaining depth integrated carbon stocks and carbon density across depth intervals, with stocks more than twice as his high in fluvial settings (mean ± SE: 226.09 ± 12.37) relative to marine settings (104.54 ± 7.11; $F_{1,136}=21.79$; $P<0.001$). This finding contrasts with previous assessments in the region which have placed vegetation (rush versus succulent/grass) as the key predictor of saltmarsh carbon stocks (Lovelock et al., 2013; Saintilan et al., 2013). These previous studies, however, have been limited in size (six sites and five sites, respectively), and confounded by different vegetation species inhabiting study sites in different estuarine settings. By using a factorial experimental design encompassing nine estuaries, we have been able to demonstrate an overarching role of geomorphic setting and sedimentary properties, rather than vegetation factors.

Second, our results show that sediment grain type is of primary importance in predicting saltmarsh carbon stocks. Our simple categorisation of sediment samples as sand, fine or mixed sediment classes was sufficient to reveal broad differences in carbon stocks according to sediment type in both fluvial and marine geomorphic settings (Figure 2). Secondary predictors of carbon density were surface vegetation structure (among sand sediments only), vegetation cover (among mixed sediments only) and sediment depth (among fine sediments only).

The role of vegetation in carbon accumulation

Interestingly, our study showed no difference in carbon stocks in the surface metre of rush versus non-rush communities. When considering the potential role of vegetation structure and composition on carbon sequestration capacity, however, it is important to note the distinction between approaches which quantify carbon stocks versus those which quantify carbon accumulation rates. In our study we are most interested in identifying sites which, if disturbed or inadequately managed, may result in the release of significant amounts of stored C. We have therefore considered only sediment carbon stocks. Carbon accumulation studies on the other hand point to the rate at which carbon is added to sediment stocks (usually the surface layers only). Whilst our results show that vegetation structure does not have a primary influence on saltmarsh carbon stocks, it may have importance to carbon accumulation rates either through differences in belowground production, aboveground litterfall or the plant's capacity to trap allochthonous materials. In fact, when compared to global estimates (Duarte et al. 2013), previous studies have shown relatively high carbon accumulation rates in rush saltmarshes and low accumulation rates in succulent/grass saltmarshes (Lovelock et al. 2013, Saintilan et al. 2013). Those findings show that the influence of vegetation factors on saltmarsh carbon sequestration should not be discounted.

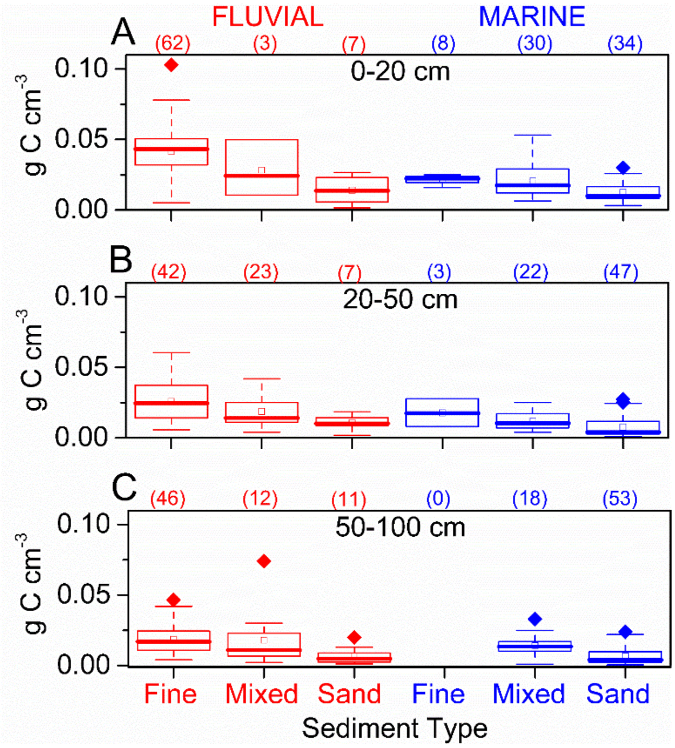

Figure 2. Boxplots of carbon density according to sediment grain type among surface (0-20 cm) (A), mid (20-50 cm) (B) and deep (50-100 cm) (C) sediments. Dark horizontal lines represent the median, open squares represent the mean, the box represents the 25th and 75th percentiles, the whiskers represent the maxima and minima, and outliers represented by filled diamonds. Samples numbers are presented in parentheses for each case.

Saltmarshes as important carbon stores

Our study shows that on a per hectare basis, NSW saltmarshes contain similar carbon stocks to saltmarshes globally. The aerial extent of saltmarshes in NSW, however, is modest (7,259 ha; Creese et al. 2009) when compared to many other temperate coastlines. This is due in part to the broader geomorphologic setting of NSW (with its relatively narrow coastal zone and the absence of broad intertidal zones in larger drowned valley estuaries) as well as significant human-induced declines in extent. Simple projection by multiplying mean carbon stock in our study (164.45 Mg C ha^{-1}) by statewide saltmarsh extent suggests NSW saltmarshes may hold 1.2 million tonnes (Mg) or more of carbon. Future efforts to delineate specific saltmarsh areas according to geomorphic zone and sediment types, will improve this statewide estimate.

It has been estimated that NSW has lost up to 70% of its coastal wetlands since European settlement (Zann 2000). Wetland draining through the construction of levees and floodgates would have been the major cause of saltmarsh loss during the 19th and early 20th centuries as floodplain agriculture expanded (especially along the northern NSW coast). If a 70% loss is valid specifically for saltmarshes, and loss of 100% of carbon from the surface 1 m upon conversion is assumed, then projection statewide from the current study suggests up to 2.8 million tonnes of carbon may have been mobilised and potentially lost to the atmosphere. Whilst further research is needed regarding the proportion and fate of carbon lost after saltmarsh conversion, it is possible that more carbon may have been mobilised through historical habitat loss relative to carbon still retained within existing saltmarsh.

Mangrove encroachment and carbon storage

As well as altering ecosystem structure, the vegetation shift from saltmarsh to mangrove also brings about a significant change in ecosystem function as demonstrated by significant increases in above and below ground carbon stocks at both sites in our study. Aboveground biomass increased across the 70 y chronosequence (Figure 3) by 130 ± 18 Mg km^{-2} y^{-1} at the polyhaline Georges River site and 52 ± 10 Mg km^{-2} y^{-1} at the marine Towra Point site - a 2.5 fold difference between sites. Assuming a carbon content of 44.6% for aboveground mangrove components – which vary from 43.7% to 44.9% and 45.2% for *A. marina* leaves, branches and stems (Alongi et al. 2003) – these biomass values equate to carbon storage increases of 58 ± 8 Mg C km^{-2} y^{-1} for Georges River and 23 ± 4 Mg C km^{-2} y^{-1} for Towra Point.

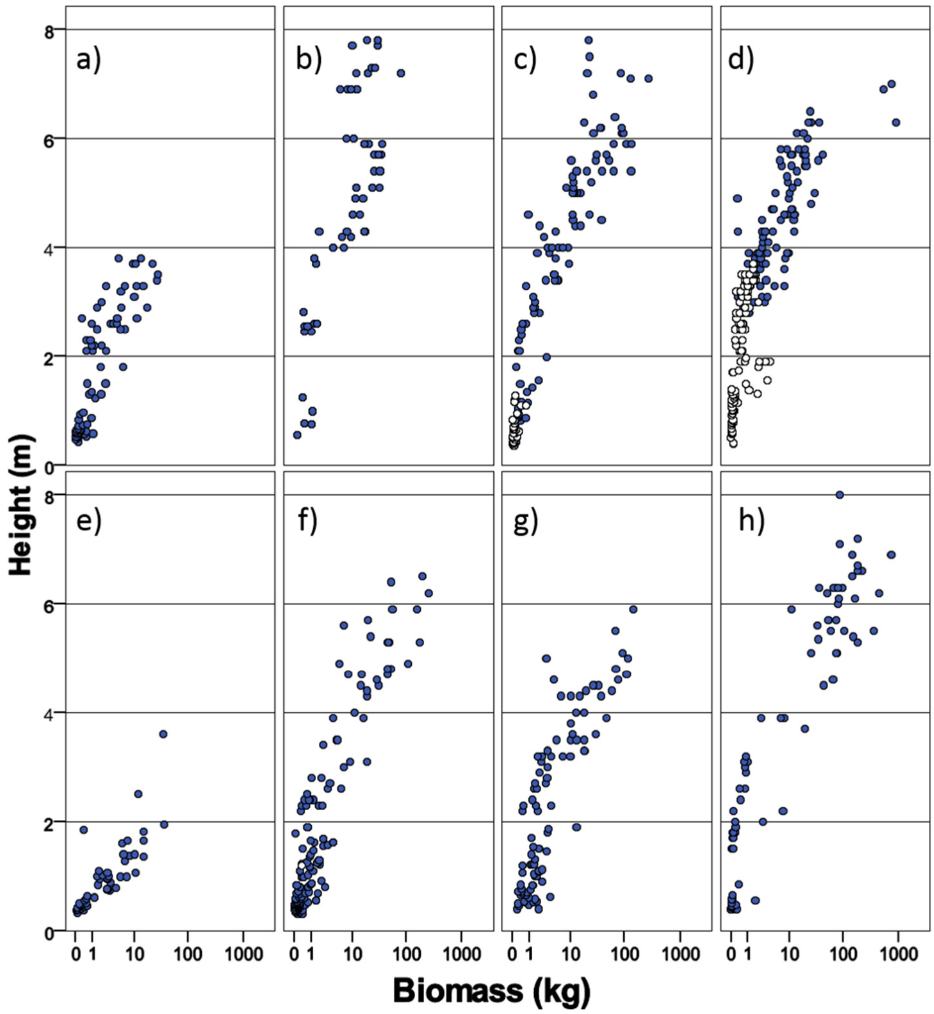

Figure 3. Scatterplots of individual tree height (tallest stem height) and derived above-ground biomass of the two mangrove species *Avicennia marina* (filled circles) and *Aegiceras corniculatum* (white cirles) across. Data are displayed by mangrove stand age for Georges River – 0-15y (a); 31-43y (b), 58-70y (c), >70y (d) - and Towra Point – 0-14y (e), 30-43y (f), 57-70y (g), and >70y (h). Note the logarithmic scale on the x axis.

Belowground stocks (inclusive of root biomass) are the most significant carbon pool in coastal wetlands and, due to their potential for long-term storage, are the primary reason for interest in coastal carbon initiatives. At both sites, and across all vegetation categories in the mangrove encroachment study, belowground carbon (quantified in the surface 100 cm) considerably outweighed aboveground carbon stocks. Further, belowground carbon store at Georges River increased by approximately 230 ± 62 Mg km^{-2} y^{-1} as mangroves replaced saltmarsh representing an increase 1.7 times faster than aboveground biomass during the same period at this site. While belowground carbon was observed to increase throughout the chronosequence at Towra Point, the relationship with mangrove age over the period 0-70 y was not as strong as for Georges River.

Research and Management

Despite their biodiversity and ecosystem service values, land use changes and habitat fragmentation continue to cause the loss of saltmarsh wetlands and their ecosystem services in Australia (Kelleway et al. 2009) and globally (Adam 2002). The ability of coastal ecosystems to sequester significant amounts of carbon is pointing the marine conservation community toward carbon credits as a potential management as well as financing tool (Lau 2012). The outcomes of our studies suggest that planning policies and on-ground activities which reverse the historical decline of coastal saltmarsh and facilitate upslope migration of both saltmarsh and mangrove in response to sea level rise will increase carbon storage in coastal wetlands (Figure 4), presenting a negative feedback to global warming. Our broad study along the NSW coastline shows carbon storage is greatest in saltmarshes dominated by fine-grained sediments and subject to fluvial inputs. If avoiding carbon emission is an objective then conservation and management intervention should be prioritised for these saltmarsh types, as their loss or degradation is likely to lead to the most substantial losses of carbon. Maintaining or enhancing the conditions which have created these carbon hotspots – such as the continued delivery of fluvial sediments and stable, terrestrial OM inputs – will also promote future C sequestration. Further research efforts will improve our understanding of the processes driving carbon sequestration in coastal habitats and help to inform regional and global carbon management and potential carbon offset schemes.

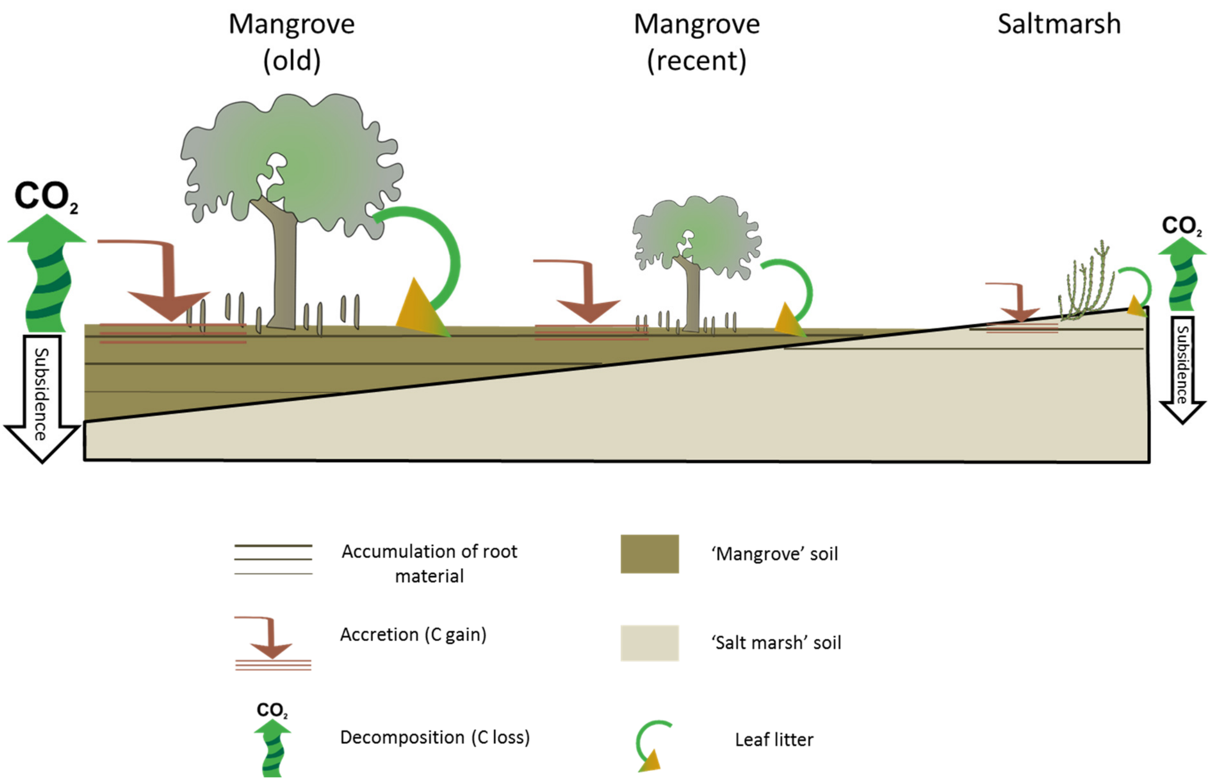

Figure 4 - Conceptual diagram of carbon dynamics of mangrove encroachment into saltmarsh. As mangroves mature, belowground production not only increases carbon storage, but may also build surface elevation, which may allow mangroves to keep pace with rising sea level.

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