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Surface evolution and carbon sequestration in disturbed and undisturbed wetland soils of the Hunter estuary, southeast Australia

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ABSTRACT

The aim of this work was to quantify the soil carbon storage and sequestration rates of undisturbed natural wetlands and disturbed wetlands subject to restriction of tidal flow and subsequent rehabilitation in an Australian estuary. Disturbed and undisturbed estuarine wetlands of the Hunter estuary, New South Wales, Australia were selected as the study sites for this research. Vertical accretion rates of estuarine substrates were combined with soil carbon concentrations and bulk densities to determine the carbon store and carbon sequestration rates of the substrates tested. Relationships between estuary water level, soil evolution and vertical accretion were also examined. The carbon sequestration rate of undisturbed wetlands was lower (15% for mangrove and 55% for saltmarsh) than disturbed wetlands, but the carbon store was higher (65% for mangrove and 60% for saltmarsh). The increased carbon sequestration rate of the disturbed wetlands was driven by substantially higher rates of vertical accretion (95% for mangrove and 345% for saltmarsh). Estuarine wetland carbon stores were estimated at 700–1000 Gg C for the Hunter estuary and 3900–5600 Gg C for New South Wales. Vertical accretion and carbon sequestration rates of estuarine wetlands in the Hunter are at the lower end of the range reported in the literature. The comparatively high carbon sequestration rates reported for the disturbed wetlands in this study indicate that wetland rehabilitation has positive benefits for regulation of atmospheric carbon concentrations, in addition to more broadly accepted ecosystem services.

1. Introduction

Wetlands are known to be important carbon sinks. Although occupying only about 5% of the Earth's surface, wetlands contain around 40% of global soil organic carbon, which is estimated at 1400 Pg C (Mitsch and Gosselink, 2000). Much of this carbon is sequestered as peat, in sediment and in plant biomass (Bridgham et al., 2006). In many wetland environments, the positive contribution to reducing atmospheric carbon concentrations resulting from soil carbon sequestration is offset by methane emissions from plant decomposition (Whiting and Chanton, 2001). A notable exception is estuarine wetlands, which are remarkably good at storing carbon with a minimal release of greenhouse gases due to the inhibition of methanogenesis by sulfates (Magenheimer et al., 1996). In addition, they are effective sinks for riverine organic soil carbon and fine-grained sediment (Meade, 1972). Overall, estuarine wetlands have a carbon sequestration capacity per unit area of about an order of magnitude higher than other wetland systems (Bridgham et al., 2006) and, if left undisturbed, can store carbon for millennia (Roulet, 2000).

Reclamation of wetlands for agriculture and urban development has substantially reduced soil carbon stores worldwide. US studies show that conversion of wetlands to cropping and pasture lands and burning of peat for fuel can reduce carbon stores by up to 50%, mostly within the first decade following land use change (Armentano and Menges, 1986). In Australia, it is estimated that 50% of wetlands have been converted to other uses since European settlement (Commonwealth Government of Australia, 1997). Coastal wetlands of New South Wales (NSW), Australia, have been affected by extensive modifications to tidal flows. In excess of 4000 impediments to tidal flow have been identified, of which approximately one third have potential for estuarine wetland rehabilitation (Williams and Watford, 1997). Even though it is not completely clear how this potential rehabilitation would affect carbon sequestration, successful rehabilitation projects in the US indicate some recovery of the carbon stores of tidal wetlands following reintroduction of tidal flows (Craft, 2001).

As carbon sequestration in estuarine wetlands is strongly linked to sediment accumulation, the rate of carbon sequestration may be

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readily determined as the product of soil organic carbon density and the rate of soil vertical accretion (Chmura et al., 2003). Soil vertical accretion contributes to wetland soil surface elevation evolution (the other major component is subsurface processes) and is one of the mechanisms by which wetlands develop and adapt to changes in estuary water levels (Kirby, 1992; Cahoon et al., 1995; Reed, 1995). If soil surface evolution is unable to match rises in estuary water levels and relatively steep landward topography precludes the landward transgression of the vegetation communities, estuarine wetland may be completely submerged (Reed, 1990; Woodroffe, 1990). Conversion of estuarine wetlands to either subtidal substrates or adjacent terrestrial communities reduces carbon sequestration, as these substrates have lower sequestration capacity (Chmura et al., 2003); furthermore, if this conversion was associated with soil oxidation or erosion, there is potential for former estuarine wetland carbon stores to be released to the atmosphere.

In this paper we examine soil carbon sequestration and surface evolution at a natural and a rehabilitated estuarine wetland, and investigate the potential effects of increasing estuary water levels. The study was conducted in the Hunter estuary, NSW, southeast Australia, in wetlands comprised of saltmarsh and mangrove habitats. These wetlands are characterized by temperate climate, small tidal range, low sediment supply and small stream discharge. No previous studies of carbon sequestration in estuarine wetlands of this type have been reported in the literature. The closest relevant information are few data on Australian wetland soil carbon concentrations (Webb, 2002 and references therein) and carbon burial rates in tropical mangrove forests of northern Australia (Alongi, 2002; Brunskill et al., 2002) and southeast Asia (Fujimoto et al., 1999; Alongi et al., 2001, 2004). Besides differences in climate and predominant species, these burial rates can be considered historic (over 100 or more years) since they have been derived from radionuclide measurements. In contrast, the soil surface evolution method of Cahoon et al. (2002) adopted in this study provides contemporary rates (01–10 years), which are more appropriate for our period of analysis. Contemporary rates may not equate to historic rates, particularly after disturbances like the ones analyzed in this paper.

2. Methods

2.1. Study sites

Field work was conducted at two sites on Kooragang Island in the Hunter estuary between November 2004 and June 2008 (Fig. 1). The Hunter River has a barrier estuary with a semi-diurnal tidal regime and a tidal range of about 1.9 m. The estuary contains around 3000 ha of estuarine wetlands dominated by mangrove and saltmarsh. Over the period from 1954 to 1994, the area occupied by saltmarsh has declined, primarily due to industrial development, drainage works and upslope migration of mangroves, while the area of mangrove has increased. The net effect has been a 30% decline in the combined area of these estuarine habitats (Williams et al., 2000).

Two sites were investigated, one disturbed (Site 1) and one undisturbed (Site 2). Tidal flows to Site 1 have been considerably modified by agricultural and urban infrastructure, and more recently by wetland rehabilitation activity. Flows to Site 1 were initially restricted by culverts, pipelines and roads constructed *ca.* 1960. In 1995 the culverts in one of the inlet channels were removed as part of a program to improve wetland condition and function. Prior to culvert removal most of the site was occupied by terrestrial pasture, saline tidal pools and degraded saltmarsh. Conversely, estuarine vegetation at Site 2 has not been directly affected by urban and agricultural activity and is undisturbed compared to Site 1. Both sites had comparable tidal influences during the study period.

Estuarine habitats of Site 1 were comprised of mangrove forest, permanent tidal pools, saltmarsh and saltmarsh pannes. Mangrove forest was characterized by dense, monospecific stands of the Grey Mangrove, *Avicennia marina* (Forsk.). Vierh. Saltmarsh was dominated by Samphire (*Sarcocornia quinqueflora* Bunge ex Ung.-Sternb) and Saltwater Couch (*Sporobolus virginicus* (L.) Kunth). Mangrove occurred lowest in the tidal frame, followed upslope by permanent tidal pools, saltmarsh pannes and saltmarsh. Saltmarsh pannes and tidal pools were either unvegetated or covered by dense stands of benthic algal matting or filamentous algae, respectively (Howe, 2005). At Site 2, mangrove forest and saltmarsh were dominated by the same species as were found at Site 1; however, saltmarsh was generally devoid of saltmarsh pannes and tidal pools.

2.2. Surface evolution and water level data

In this study measurement of soil surface evolution was disaggregated into changes in elevation and changes in vertical accretion after the method of Cahoon et al. (2002). This allowed the contribution to surface elevation change due to surface accretion processes to be separated from that due to subsurface processes such as shallow subsidence, water table fluctuations and root accumulation. While the amount of vertical accretion is required to calculate wetland carbon sequestration rate, it is the absolute change in surface elevation that determines whether a wetland is able to keep pace with rising sea level. To investigate this last point, hydraulic data were also gathered.

Change in soil surface elevation and vertical accretion were determined by establishment of surface elevation tables (SETs) and feldspar marker horizons after the method of Cahoon et al. (2002). Six SET sites were established at Site 1 for this study in March 2005: three in mangrove habitat near the Hunter River and three in saltmarsh habitat approximately 350 m north of the mangrove sites (Fig. 1). A further six SET sites (three saltmarsh and three mangrove sites) were established at Site 2 by the Australian Catholic University in January 2002 (Fig. 1). Site 2 has been sampled since November 2004, and previous data (January 2002–November 2003) were provided by K. Rogers. To establish the SETs, a 6 m steel post was driven vertically into the marsh until refusal (approx. 4.5 m) to form a benchmark at each site. A platform was constructed around the benchmark to allow access without damaging the marsh surface. A horizontal arm, fitted with a plate and nine nylon rods, was mounted on the benchmark post and leveled. The arm was able to be rotated to any of eight fixed compass bearings, which allowed the same section of marsh to be repeatedly sampled over time. Surface elevation was recorded at four bearings at each SET. At each of the selected bearings, the rods were gently lowered to the marsh surface and the height of the rods above the plate recorded and averaged. Change in soil vertical accretion was determined by establishment of feldspar marker horizons at 36 locations: 18 in mangrove and 18 in saltmarsh. Half of the sites were located at Site 1 and the remainder at Site 2. Feldspar was laid directly onto the marsh surface within a 0.25 m^2 area. The depth of sediment deposited on top of the feldspar was recorded by extracting a small core from each square. The depth of accretion above the marker was measured to the nearest millimeter with a ruler at three positions and averaged.

Vertical accretion and surface elevation data were regressed against water level data. Estuary water level data for the lower Hunter were obtained from Manly Hydraulics Laboratory's Ironbark Creek gauge, located approximately 2 km southwest of the study area in the south arm of the Hunter River (Fig. 1). All water level