Contemporary Rates of Carbon Sequestration Through Vertical Accretion of Sediments in Mangrove Forests and Saltmarshes of South East Queensland, Australia

Author

Lovelock, Catherine E, Adame, Maria Fernanda, Bennion, Vicki, Hayes, Matthew, O'Mara, Julian, Reef, Ruth, Santini, Nadia S

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4 **Contemporary rates of carbon sequestration through vertical accretion of sediments in mangrove**

5 **forests and saltmarshes of South East Queensland, Australia**

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7 Catherine E. Lovelock, Vicki Bennion, Maria Fernanda Adame, Matthew Hayes, Julian O'Mara, Ruth

8 Reef, Nadia S. Santini.

9

10 The School of Biological Sciences

11 The University of Queensland

12 St Lucia, Queensland QLD 4072

13 Australia

14

15 T +61 7 33652304

16 F +61 7 3365 4755

17 Email c.lovelock@uq.edu.au

18

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21   **Abstract**

22   Mangrove forests and saltmarshes are important habitats for carbon (C) sequestration in the coastal

23   zone but variation in rates of C sequestration and the factors controlling sequestration are poorly

24   understood. We assessed C sequestration in Moreton Bay, South East Queensland in mangrove

25   forests and saltmarshes that span a range of environmental settings and plant communities,

26   including mangrove forests and tidal marshes on the oligotrophic sand islands of the eastern side of

27   Moreton Bay and on the nutrient enriched, western side of the bay adjacent to the city of Brisbane.

28   We found that rates of C sequestration in sediments were similar among mangrove forests over the

29   bay, despite large differences in the C density of sediments, because of different rates of vertical

30   accretion of sediments. The C sequestration on the oligotrophic sand island tidal marshes,

31   dominated by *Juncus kraussii,* had the highest rate of C sequestration in the bay while the western

32   saltmarshes, which were dominated by *Sarcocornia quinqueflora,* had the lowest rate of C

33   sequestration. Our data indicate C sequestration varies among different tidal wetland plant

34   community types, due to variation in sediment characteristics and rates of sediment accretion over

35   time.

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Introduction

Mangrove forests and saltmarshes have large amounts of carbon (C) stored within their sediments

(Chmura et al. 2003; Mcleod et al. 2011). High levels of C accumulate in these tidal wetland habitats

because sediments accrete over time (add volume) in response to rising sea levels due to inputs of

sediments and detritus (e.g. McKee et al. 2007; McKee 2011). The detritus in mangrove forest and

saltmarsh sediments can be autochthonous, comprised predominantly of roots (Middleton and

McKee 2001) but also allochthanous, with contributions from seagrass, terrestrial soils and

macroalgal sources (Middelburg et al 1997; Bouillon et al. 2003; Adame et al. 2012). Decomposition

of material is impeded by flooded, anoxic sediment conditions thereby facilitating C sequestration

(Mcleod et al. 2011).

The large stocks of carbon within sediments and the potential for its emission once these habitats

are disturbed (Lovelock et al. 2011) has resulted in proposals to introduce mangrove forests for

their C values into conservation schemes such as Reducing Emissions from Deforestation and

Degradation (REDD+) (Siikaamaki et al. 2012). Additionally, the accumulation of C within sediments

of mangrove forests is a means of mitigating increasing C dioxide concentrations in the atmosphere,

thereby providing an incentive for restoration and improved management of coastal wetlands in

order to increase C sequestration.

Most estimates of C stocks and rates of accumulation have come from work with sediment cores,

where rates of accumulation are estimated based on dating of accumulated sediment C over long

periods of time (e.g. Chmura et al. 2003). While knowledge of past rates are essential for

establishing the potential of mangrove forests and salt marshes to sequester C they do not inform us

of current rates of sequestration because over time, vegetation, hydrology and levels of nutrient

availability may change, all of which will affect rates of C sequestration (Alongi 2012). Current rates

of C sequestration in coastal habitats can be estimated from restoration or natural chronosequences

(e.g. Alongi et al. 2008; Lovelock et al. 2010; Osland et al. 2012). Additionally, measures of sediment

66   surface elevation change in conjunction with C density of sediments can be used to estimate rates of

67   C sequestration (e.g. Howe et al. 2009; Rogers et al. 2013).

68   Some of the main factors that are likely to influence C sequestration in coastal wetland habitats are

69   geomorphic setting, plant species composition and nutrient availability. Geomorphic settings

70   influence mangrove forests and saltmarsh through their effects on a range of factors, including the

71   type and abundance of sediments and nutrient availability. Estuarine systems often have a higher

72   availability of fine sediments, nutrients and allochthonous materials that can be incorporated into

73   tidal wetland sediments compared to other settings (e.g. oceanic) (Kristensen et al. 2008; Adame et

74   al. 2010; Adame et al. 2012). In estuarine settings, fine sediments and nutrients lead to high rates of

75   primary production of mangrove forests (Lovelock et al. 2007) which can lead to stimulation of root

76   growth and increased rates of deposition of root detritus to sediments (McKee 2011), but sediments

77   may also dilute autochthonous C additions (Mcleod et al. 2011). Additionally, high contributions of

78   allochthonous organic matter may prime decomposition pathways (Bianchi 2011), which may

79   increase microbial oxidation of organic matter and thus reduce C sequestration. High nutrient

80   concentrations may also reduce allocation of C to roots and root detritus and stimulate

81   decomposition (McKee et al. 2007).

82   In non-estuarine systems, such as those in oceanic settings, associated with reefs or sand cays and

83   islands, mineral sediments may also contribute to sediment volume (e.g. Lovelock et al. 2011; McKee

84   et al. 2007). But organic matter composition of mangrove sediments in oceanic settings has been

85   observed to be largely comprised of plant roots indicating that plant production, even if it is low

86   compared to estuarine systems, can be the most important contributor to sediment C sequestration

87   (McKee et al. 2007; McKee 2011; Adame et al. 2013). Extremely low nutrient availability in

88   hydrologically isolated karstic settings, for example landward scrub forests, may result in such low

89   levels of primary production that C sequestration may be low compared to areas with more regular

90   tidal flushing (McKee et al. 2007; Adame et al. 2013).

91   The plant species composition may also strongly influence C sequestration in coastal vegetation. In

92   Caribbean peat sediments in mangrove forests are comprised of roots of *Rhizophora mangle* with

93   limited contributions from *Avicennia germinans* (McKee et al. 2007). The highly organic sediments in

94   the Indo-West Pacific are also associated with species within the family Rhizophoraceae (Donato et

95   al. 2011; Kauffman et al. 2011). Some saltmarsh species with highly developed rhizomes are

96   associated with highly organic sediments, for example *Spartina alternifora* in the USA (Chmura et al.

97   2003) and *Juncus kraussii* in Australia (Congdon and McComb 1980) while others, for example

98   communities of chenopods that occupy hypersaline sediments, may provide limited contributions to

99   sediment C. Additionally, different species of mangrove trees with their varying root morphologies

100   lead to different rates of increases in sediment volume, reflecting their belowground root growth

101   and also their capacity to trap and bind sediments (Krauss et al. 2003).

102   Here we used an array of surface elevation tables that measure increases in sediment volume over

103   time in conjunction with measurements of C density of surface sediments to estimate current rates

104   of C sequestration in a large subtropical embayment, Moreton Bay, South East Queensland. The Bay

105   has both mangrove forests and saltmarsh communities over a gradient in nutrient availability. We

106   test the following hypotheses:

107   1. Rates of C sequestration differ among habitat types (mangrove vs saltmarsh) and with

108   variation in plant communities (mangrove and saltmarsh species).

109   2. Rates of C sequestration are sensitive to nutrient supply, with declining rates of C

110   sequestration at high levels of nutrient enrichment.

111

112   **Materials and Methods**

113   *Site description*

114   Moreton Bay is a large semi-open embayment on the east coast of Australia. It is bound on the

115   eastern side by sand islands that reach approximately 200 m in elevation and by a deltaic coast on

116   the western side in which five rivers flow from their catchments to the bay. The city of Brisbane

117   (population 3.5 million) resides on the western side of Moreton Bay. The climate is sub-tropical and

118   thus the bay is fringed by mangrove forests low in the intertidal zone with saltmarsh and

119   cyanobacterial mats in the high intertidal zone (Lovelock et al. 2011).

120   The mangrove forests of Moreton Bay are dominated by *Avicennia marina* on the western side of

121   the Bay, but have a high abundance of *Rhizophora stylosa* in the eastern Bay (Dennison and Abal

122   1999). Saltmarsh communities are more variable in composition than mangrove forests. Where

123   fresh water is abundant in the high intertidal, as it is in the eastern bay and in some locations in the

124   west, the rush *Juncus kraussii* dominates while on the hypersaline high intertidal salt flats of the

125   western bay communities are dominated by chenopod species from the genus *Sarcocornia* and

126   *Suaeda*.

127   South East Queensland is subtropical and has annual variations in climate. Mean air temperature is

128   highest from October to April (mean minimum temperature is 18 °C, mean maximum temperature

129   26 °C) and low in May-September (mean minimum temperature is 8 °C, mean maximum

130   temperature 21 °C). Rainfall is highest in the warmer summer months from November to March

131   (150 mm/month), with another smaller peak in rainfall in May (200 mm/month). Rainfall influences

132   river flows, while wind speed affects resuspension of sediments in the bay through wave action (You

133   2005).

134   *Experimental design*

135   Three sites were chosen on both the eastern and western bay that comprised both mangrove and

136   saltmarsh habitats (Figure 1). On the western bay, the three sites were located within the Tinchi

137   Tamba Wetlands Reserve (Brisbane City Council), Nundah Creek (Boondall Wetlands Park) and in the

138   south at Halloran Reserve (Redlands Shire Council). At Tinchi Tamba and Nundah Creek the

139   mangrove was comprised of *A. marina* of about 10 – 13 m in height. At Halloran Reserve *A.marina*

140   was dominant but *Rhizophora stylosa* was also abundant. The saltmarsh in all sites was dominated

141   by *Sarcocornia quinqueflora* with *Suaeda australis* also being present in most plots. In the eastern

142   bay three sites were chosen along the western side of North Stradbroke Island. Two sites were north

143   of the town of Dunwich, between Myora Light and Amity Point (Amity North and Amity South), while

144   one was south of Dunwich, to the south of Adams Beach. The mangrove forests were approximately

145   10 – 15 m in height and were comprised of both *A. marina* and *R. stylosa*. The eastern saltmarsh was

146   dominated by *Juncus kraussii*.

147   At each of the sites, three plots at least 30 m apart, were established and rod surface elevation

148   tables and marker horizons (RSET-MH) were installed. Within the mangrove forests, plots were

149   within 20 - 50 m of the creek bank or seaward edge. An additional three plots with RSET-MH were

150   installed in the adjacent landward saltmarsh. In total, 36 RSET-MHs were installed throughout

151   Moreton Bay. Detailed description of increment in surface elevation and local rate of sea level rise

152   within Moreton Bay are available in Lovelock et al. (2011). The position of each RSET-MH in the

153   mangrove forests were determined by measuring the depth of the water covering the site at high

154   tide using dyed cotton strips (English et al. 2000). Sites on the eastern bay were lower in the

155   intertidal than those on the western bay with a position of 1.36 ± 0.01 m relative to the lowest

156   astronomical tide (LAT) in the eastern bay compared to 1.87 ± 0.06 m relative to LAT in the western

157   bay. Saltmarsh sites are all infrequently inundated and are positioned more than 2.25 m above LAT.

158   *Measurements of surface elevation increments*

159   Changes in the surface elevation of wetlands were measured using the rod surface elevation table –

160   marker horizon (RSET-MH) approach developed by Cahoon et al. (2002a and b). RSETs were installed

161   between February and June of 2007. The depth of the benchmark varied between 4 – 17 m, with a

162   mean of 12 m. A portable aluminum plank was used to access the benchmark rod, attach the RSET

163   and measure the surface elevation without disrupting surface sediments. Once the plank is in place

164   and the RSET is attached to the base, the measuring arm is leveled horizontally. Nine fibre glass pins

165   are then lowered to the surface. Elevation is measured as the amount of pin protruding above the

166   measuring arm using a ruler with precision of 1 mm, where the change throughout the

167   measurement interval is the change in pin length with reference to the base depth. The device has a

168   range of fixed measurable positions around the rod. In this study, we used four measuring positions

169   at 90° to each other to give 36 measurements for each plot. A mean of 36 measurements was

170   calculated for each replicate RSET. For the first 18 months of the study, RSET-MHs were measured

171   every 3-4 months, after which sampling intervals were lengthened to approximately 6 months to

172   cover the winter dry (April – November) and summer wet seasons (November – April). Rates of

173   surface elevation change (in mm y^{-1}) are the slopes from the regression of surface elevation vs. time

174   from June 2007 until November 2011.

175   *Sediment sampling*

176   A known volume of surface sediments was sampled to 5 cm depth in July 2011 using a modified 50

177   ml plastic syringe. Three replicate sediment samples were collected from each experimental plot. To

178   determine bulk density known volumes of sediment were dried at 60°C until samples reached a

179   constant weight and then weighed. Dry bulk density was calculated as dry weight/volume. Samples

180   were ground in a ball mill and then analysed for total C and nitrogen by combustion (LECO). Total

181   sediment phosphorus was analysed using ICPMS. All analyses were performed at the Analytical

182   Services Laboratory at the University of Queensland.

183   *Data analyses*

184   The C sequestration (g C m^{-2} y^{-1}) was calculated as the rate of surface elevation change (mm y^{-1})

185   multiplied by the C density of the sediment. Analysis of the variation in %C of sediments, bulk

186   density , C sequestration and sediment nutrients was by generalized linear models where side of the

187   bay (east/west) and habitat (mangrove/saltmarsh) were fixed factors in the model and where site

188   was nested (and random) in eastern/western side of the bay. The relationship between C

189   sequestration and sediment nutrient concentrations and molar ratios was tested using regression

190   analysis. Data was log transformed prior to analysis to normalize the variance of the data. Normality

191   was assessed by inspecting residual plots. Analyses were performed using Data Desk (version 6.3,

192   Data Descriptions Ithaca, NY, USA).

193

194   **Results**

195   In the eastern bay % C per dry mass of sediment was highest in the *Juncus* saltmarsh habitat (mean

196   15.1 ± se 3.0 %) compared to mangrove sediments (mean 1.3 ± se 0.3 %) (Fig. 2). In contrast, in the

197   western bay mangrove sediments had higher %C (8.4 ± se 0.5 %) than the *Sarcocornia* dominated

198   saltmarsh (mean 2.2 ± 0.7%). These opposing patterns of %C in mangrove forest and saltmarsh

199   sediments on either side of Moreton Bay gave rise to a highly significant bay x habitat interaction in

200   the statistical model ($F_{1,4}$ = 96.2, P = 0.0006).

201   Dry bulk density also varied between mangrove forests and saltmarsh, but in different ways

202   depending on the location within the bay (Fig. 3; bay x habitat, $F_{1,4}$ = 76.12, P = 0.0001). Bulk density

203   was higher in the mangrove forests on the eastern side of the bay (0.98 ± 0.02 g cm^{-3}) compared to

204   the *Juncus* saltmarsh (0.33 ± 0.09 g cm^{-3}). In the western side of the bay mangrove forest sediments

205   had lower BD (0.49 ± 0.02 g cm^{-3}) than *Sarcocornia* saltmarsh (1.12 ± 0.08 g cm^{-3}).

206   The density of C (product of BD and % C) was highest in *Juncus* saltmarsh habitats in the eastern bay

207   (bay x habitat interaction $F_{1,24}$ = 32.4, P = 0.005) and lowest in the mangrove forests in the eastern

208   side of the bay (Fig. 4). Mangrove forest sediments on the western side of the bay had a C density

209   that was approximately double that of mangrove forests in the eastern side of the bay.

210   Surface elevation change per year varied significantly among sites ($F_{4,21}$ = 5.72, P = 0.003) (Table 1).

211   Surface elevation gains tended to be higher on the eastern side of the bay than the western side of

212   the bay ($F_{1,4}$ = 7.42, P = 0.053) and higher in the mangrove forests compared to the saltmarsh ($F_{1,4}$ =

213   6.91, P = 0.058). The difference between mangrove and saltmarsh surface elevation varied over sites

214   (site x habitat interaction $F_{1, 21}$ = 5.59, P = 0.0029). Surface elevation gains in saltmarsh on the

215   eastern side of the bay were higher at Adams Beach than at Amity North and South, and in the

216   western bay were higher at Nundah Creek than in saltmarsh at Tinchi Tamba or Halloran Reserve.

217   The C sequestration rates, calculated from changes in surface elevation over time (Table 1) and C

218   density (Fig. 4) was significantly different among sites ($F_{4,23}$ = 5.98, P = 0.0019) with C sequestration

219   particularly high in *Juncus* saltmarsh at Amity North and Adams Beach in the eastern side of the bay

220   (Fig. 5). In contrast C sequestration was very low in the *Sarcocornia* dominated marshes of the

221   western bay (Fig. 5). Rates of C sequestration in mangrove forest sediments were fairly homogenous

222   over all sites (grand mean 76 ± 16 g C m^{-2} y^{-1}), although rates were low at Tinchi Tamba.

223   There was no significant relationship between rates of C sequestration and P (g per sediment

224   volume), N:P molar ratio or C:N molar ratio. Carbon sequestration varied significantly with N and C:P

225   of sediments, but the relationships were highly variable (data not shown). Carbon sequestration was

226   low with low C:P molar ratio and increased as C:P increased before declining at very high C:P ratios

227   ($F_{2,33}$ = 3.59, P = 0.039, R^2 = 0.179). Rates of C sequestration increased with N of sediments, although

228   the relationship was also variable ($F_{1,33}$ = 6.66, P = 0.014, R^2 = 0.143). However, assessment of the

229   mean sediment nutrient characteristics over the different vegetation communities indicated that

230   while P and N were particularly low in the eastern mangrove forest sediments, C:P and N:P ratios

231   were highest in the eastern saltmarsh where C sequestration was also high (Table 2). N:P ratios were

232   lower than or similar to the Redfield ratio (<16:1) in mangrove forests and western saltmarsh

233   indicating N limitation to growth, but were significantly higher (40) in eastern saltmarsh. C:P ratios

234   were between 170-576 in the mangrove forests and western saltmarsh but were over 1500 in the

235   eastern saltmarsh (Table 2).

236

237   **Discussion**

238

239   Rates of C sequestration in mangrove forests were within the range of those estimated from

240   sediment cores, although on the low end of the range (global range of 20-949 g C m^{-2} y^{-1}, mean of

241   163 g C m^{-2} y^{-1}; Breithaupt et al. 2012; Chmura et al. 2003; Mcleod et al. 2011; Ruiz-Fernández et al.

242   2011). The rates of C sequestration were similar to those observed in riverine, mineral sediment-rich

243   settings in tropical Australia (mean 180 g C m^{-2} y^{-1}, Brunskill et al. 2002) and in southern Australia

244   (105 g C m^{-2} y^{-1}, Howe et al. 2009; 256 g C m^{-2} y^{-1}, Saintilan et al. 2013). The rates are low compared

245   to those from the Caribbean region (Mexico and Florida) where sediments are highly organic

246   (Chmura et al. 2003).

247   Our estimates of C sequestration in mangrove sediments may be low due a range of physical and

248   ecological processes, including lower primary productivity in subtropical forests, low inputs of

249   allochthanous carbon sources and high rates of export (Twilley et al. 1992), but may also be due to

250   limitations to our approach for estimating C sequestration based on C concentrations in surface

251   sediments. Using C in the surface sediments may underestimate C sequestration because root

252   detritus contributes to sediment C throughout the sediment profile. For example, mangrove roots

253   tend to grow within older decomposing root structures (McKee 2001). Assessment of the amount of

254   contemporary C sequestration throughout the sediment profile will require the development of new

255   tools that can differentiate accumulation of contemporary C from prior forest (or saltmarsh)

256   contributions. More likely our approach may give rise to overestimates because C concentrations in

257   the top sediment surface layer may be higher than that incorporated into the sediment profile

258   (Breithaupt et al. 2012) as a large proportion of organic matter may be lost through diagenesis

259   within the first year of deposition (Duarte and Cebrián 1996). We sampled sediments that represent

260   more than one year of sediment deposition (top 5 cm, or 5-50 years of accumulation) to account for

261   this potential problem, but overestimation may still occur if C mineralization rates are high.

262   Additionally, C can be lost from deep in the sediment profile in porous, sandy sediments which could

263   also lead to overestimates of long term C storage (Rogers et al. 2013).

264   Rates of C sequestration in saltmarsh sediments were highly variable. Marshes dominated by *Juncus*

265   had higher rates of sequestration than the hypersaline saltmarshes dominated by *Sarcocornia* (Fig

266   5). The rates of C sequestration in the *Juncus* marshes in our study were similar to those reported for

267   more southern *Juncus* marshes in Australia (207 g C m^{-2} y^{-1}, Saintilan et al. in review) and higher than

268   rates observed in other salt marshes in North America and Europe (mean 105 ± 55 g C m^{-2} y^{-1};

269   Chmura et al. 2003). The high rates of C sequestration in *Juncus* marshes may reflect the important

270   role of plants with perennial rhizomes in promoting C sequestration in coastal marshes. In contrast,

271   C sequestration in *Sarcocornia* marshes was very low (mean 8.6 ± 4.0 g C m^{-2} y^{-1}). Values for C

272   sequestration in *Sarcocornia* marshes in Moreton Bay were much lower than those reported for

273   similar vegetation at a more southern site (137 g C m^{-2} y^{-1}, Howe et al. 2009), but within the range of

274   that reported for a number of south eastern Australian sites (mean of 46 g C m^{-2} y^{-1}, Saintilan et al.

275   2013). In the study of Howe et al. (2009) vertical accretion rates of sediments were ten-fold higher in

276   the saltmarsh in the Hunter River than they are in Moreton Bay. High vertical accretion in the Hunter

277   River may be because of high sediment availability and also that the saltmarsh is lower in the tidal

278   frame at this site than it occurs in Moreton Bay thereby favoring high sedimentation rates. In

279   Moreton Bay mangrove forests sediment accretion is higher lower in the intertidal zone (Lovelock et

280   al. 2011). Additionally, in the study of Howe et al. (2009) root contributions of rhizomes of the grass

281   *Sporobolus virginicus* may have also contributed to high rates of C sequestration. Although *S.*

282   *virginicus* is an important component of saltmarsh in South East Queensland (Traill et al. 2011), it did

283   not occur in our plots.

284 Across the bay, opposing patterns of variation in %C and bulk density of sediments gave rise to a

285 high level of homogeneity in rates of C sequestration in mangrove forests despite vastly different

286 sediment characteristics (Fig. 3 and 4). While mangrove sediments in the east had low %C and high

287 dry bulk density due to high contributions of sand, those in the west had higher %C but lower dry

288 bulk density. Additionally, surface elevation changes were lower in the west compared to the east

289 (Table 1) largely because of subsidence or compaction of the sediments in the western bay (Lovelock

290 et al. 2011), which may be related to differences in the species composition of the forests and their

291 differences in root production and root structural traits (Krauss et al. 2003), as well as due to

292 differences in sediment characteristics. In the west there are few *R. stylosa* and *A. marina* is

293 dominant, while the eastern bay has a higher abundance of *R.stylosa*.

294 Carbon sequestration was not strongly correlated with nutrient concentrations of sediments or

295 molar ratios of elements, although there was weak evidence for increasing C sequestration with

296 increasing %N and non-linear trends with C:P ratios of sediments. These highly variable trends likely

297 reflect the multiple factors that influence root production, trapping of allochthanous C and

298 decomposition of organic matter in tidal wetlands. Low rates of plant production and root

299 contributions to sediments may limit C sequestration in low nutrient habitats (McKee et al. 2007;

300 Castaneda-Moya et al. 2011; Adame et al. 2013). But low levels of nutrient availability may also

301 favour plant species that allocate a high proportion of fixed carbon to roots and that have tissues

302 that decompose slowly (Chapin 1980). Although high nutrient environments increase plant

303 production, decomposition of organic matter can also be enhanced (Quall and Richardson 2000)

304 which may lead to reductions in C sequestration. Across our study sites in Moreton Bay the lowest

305 rates of C sequestration occurred at Tinchi Tamba where C:P and C:N ratios were very low, indicating

306 nutrient enrichment. Extremely high sediment P concentrations that are caused by pollution from

307 agriculture and urban development are likely to be detrimental to C sequestration in tidal wetland

308 ecosystems.

309   The highest rates of C sequestration of sediments in Moreton Bay occurred where plants have

310   rhizomes, sediments are highly organic with relatively low bulk density, where rates of elevation

311   change are moderate and where sediment C:P ratio and C:N ratios are high. Our study indicated

312   factors that reduce surface elevation gains or C inputs, e.g. reduced sediment availability and

313   nutrient pollution are likely to reduce rates of C sequestration within Moreton Bay, while moderate

314   rates of sea level rise and reductions in nutrient pollution in Moreton Bay are likely to increase C

315   sequestration.

316

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416   **Acknowledgements**

420

421

Figure Captions

423   Figure 1. Location of study sites within Moreton Bay, Queensland, Australia (A). An example of the

424   location of the mangrove forest and saltmarsh plots within a study site, in this case Nundah Creek

425   (B). M1, M2 and M3 are within the mangrove forest (stippled), while SM1, SM2 and SM3 are located

426   within the saltmarsh (open).

427   Figure 2. Carbon concentration in sediments (% per dry mass) in mangrove forests (black) and

428   saltmarshes (grey) at six sites in Eastern and Western Moreton Bay. Values are means and standard

429   errors of 3 plots per site.

430   Figure 3. Dry bulk density of surface sediments in mangrove forests (black) and saltmarshes (grey) at

431   six sites in Eastern and Western Moreton Bay. Values are means and standard errors of 3 plots per

432   site.

433   Figure 4. Volume-specific sediment carbon content in mangrove forests (black bars) and saltmarshes

434   (grey bars) in Moreton Bay. Values are means and standard errors of 3 plots per site.

435   Figure 5. Carbon sequestration in g C m^{-2} y^{-1} in surface sediments in mangrove forests (black) and

436   saltmarshes (grey) at six sites in Eastern and Western Moreton Bay. Values are means and standard

437   errors of 3 plots per site.

438

439

440   Table 1. Mean rates of surface elevation change (± standard error, N= 3) for mangrove forest and
441   saltmarsh sites in Moreton Bay, Queensland, Australia. Negative elevation change indicates a loss of
442   elevation over time.

443

	Surface elevation change mm year^{-1}	
	Mangrove	Saltmarsh
Eastern Moreton Bay		
Amity North	5.89 ± 0.95	-0.08 ± 0.18
Amity South	7.05 ± 0.82	0.18 ± 0.61
Adams Beach	6.23 ± 1.11	5.92 ± 0.35
Western Moreton Bay		
Tinchi Tamba Reserve	0.41 ± 0.57	0.45 ± 0.32
Nundah Creek	2.34 ± 0.56	1.22 ± 0.92
Halloran Reserve	2.42 ± 1.21	0.11 ± 0.09

444

445

446 Table 2. The dominant plant species and growth form and the mean sediment nutrient

447 characteristics (± standard error, N= 9) for mangrove forests and saltmarshes on the eastern and

448 western side of Moreton Bay, Queensland, Australia. Carbon:phosphorus (C:P),

449 nitrogen:phosphorus (N:P) and carbon:nitrogen (C:N) ratios are expressed as molar ratios. Different

450 letters after the means indicate significant differences at P<0.05.

451

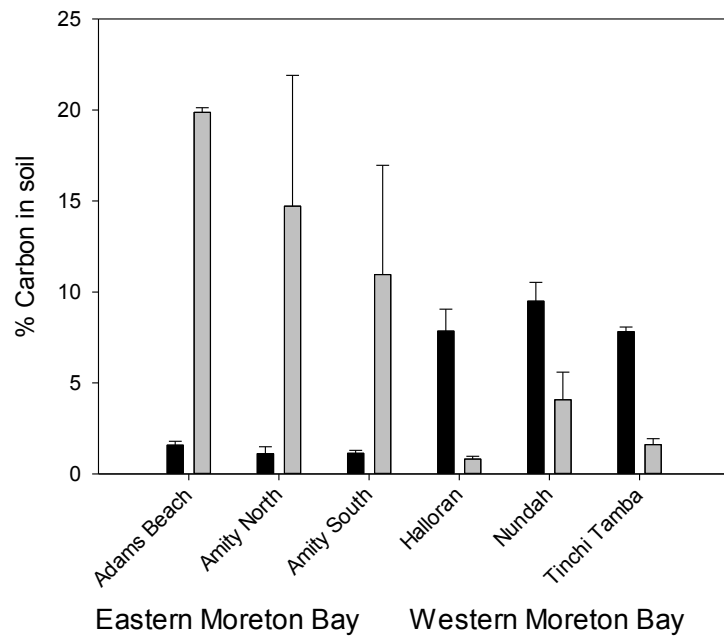| | Eastern Bay | | Western Bay | |
	Mangrove	Saltmarsh	Mangrove	Saltmarsh
Dominant species	*Avicennia marina, Rhizophora stylosa*	*Juncus krausii*	*Avicennia marina*	*Sarcocornia quinqueflora*
Plant form	tree	rush	tree	herb
Total phosphorus (mg cm^{-3})	63 ± 7 a	222 ± 74 a	356 ± 51 b	363 ± 49 b
Total nitrogen (g cm^{-3})	0.0157 ± 0.0032 a	0.272 ± 0.139 b	0.204 ± 0.027 b	0.196 ± 0.057 b
C:P	576 ± 45 a	1898 ± 512 b	392 ± 60 a	170 ± 35 c
N:P	6.35 ± 1.0 a	40.4 ± 6.7 b	15.5 ± 1.7 c	12.1 ± 2.5 c
C:N	120 ± 28 a	65 ± 21 b	25 ± 3 c	16 ± 3 c

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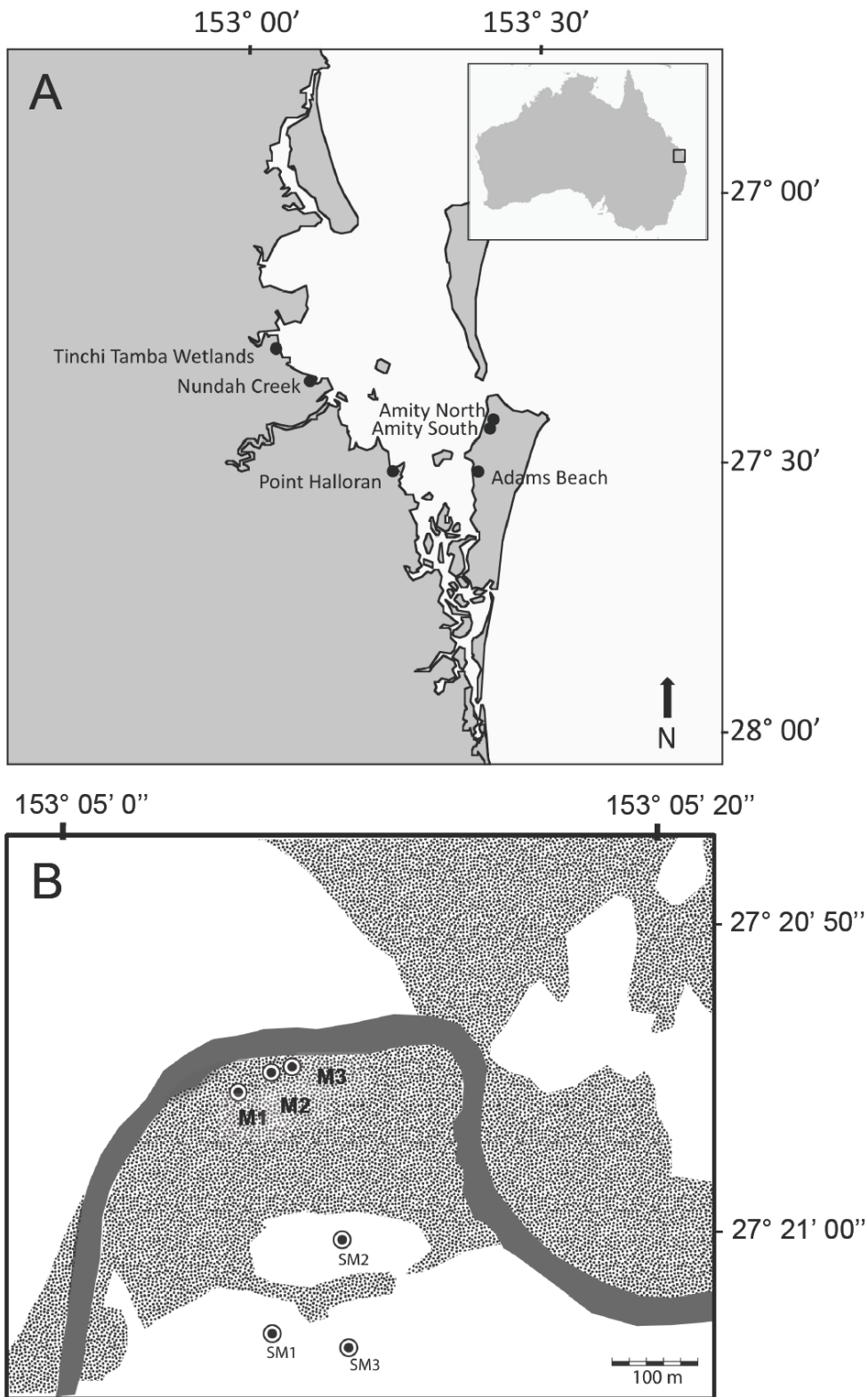
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457 Figure 1. Location of study sites within Moreton Bay, Queensland, Australia (A). An example of the
458 location of the mangrove forest and saltmarsh plots within a study site, in this case Nundah Creek
459 (B). M1, M2 and M3 are within the mangrove forest (stippled), while SM1, SM2 and SM3 are located
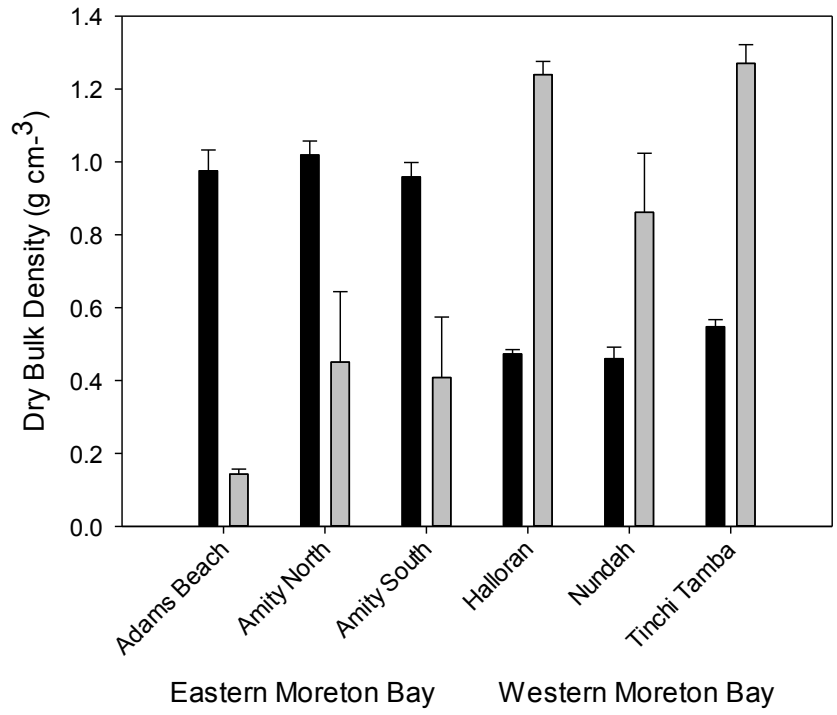460 within the saltmarsh (open).

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Eastern Moreton Bay   Western Moreton Bay
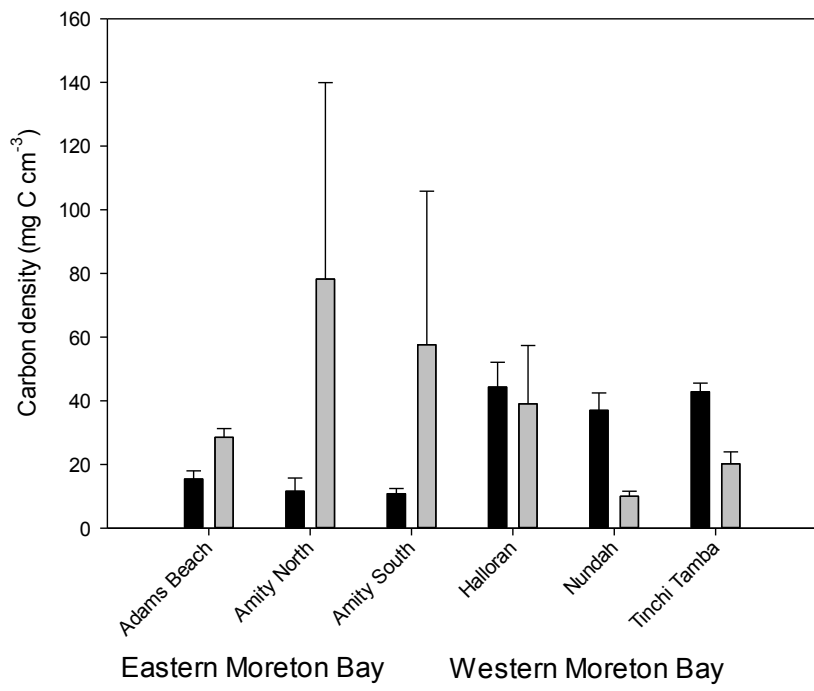
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464   Figure 2

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468 Eastern Moreton Bay   Western Moreton Bay

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470   Figure 3.

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475   Eastern Moreton Bay   Western Moreton Bay

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479 Eastern Moreton Bay   Western Moreton Bay

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