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Vertical accretion rates of mangroves in northeast Brazil: Implications for future responses and management

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ABSTRACT

Northeast (NE) Brazil has a semi-arid climate and low plant diversity mangroves confined to estuaries influenced by a meso-tidal regime (~2.5 m). Whilst these mangroves are protected by state and federal legislation, large areas have been lost to urban encroachment and shrimp farms, a situation continuing today. This work discusses the potential impacts of sea level rise on mangroves in three estuaries in NE Brazil. The three sites witness decreasing annual precipitation associated with global climate change, which will have the dual effect of increasing water and salinity stress, decreasing productivity, as well as reducing inputs of allochthonous fluvial material. Simultaneously, since the 1990's local sea level rise (3.5 mm/yr) has been greater than the global average (3.2 mm/yr). Average accretion rates, derived from ^{137}Cs and ^{210}Pb dating in fringing mangroves in this study, are likely to be insufficient to maintain surface elevation and keep pace with sea level rise. Notwithstanding, historical data suggest that there has been no recorded mangrove loss as a result of sea level rise and in some areas a recorded gain in the upper estuarine mangrove reaches. At present there are limited inputs from the watershed in all three estuaries, mostly due to decreasing annual rainfall and river damming, limiting the transport capacity and hampering the export of sediments to the fringing mangroves and the sea and limiting the ability of the fringing mangroves to respond to sea level rise. However, this has been suggested to result in the accumulation of soils along river margins further upstream forming new islands and potentially expanding estuarine mangroves upstream. Both the fringing and the estuarine mangroves are surrounded by both licensed and unlicensed settlements, as well as dunes preventing expansion into higher elevation areas as a response to sea level rise. Recent changes to legislation protecting these ecosystems put a further strain on adaptation and resilience capacity, increasing the threat to these important coastal systems and the ecosystem services they provide.

1. Introduction

Climate change has been noted as one of the greatest threats to global ecosystems due to the rate and magnitude of changes (IPCC et al., 2022), which is particularly acute for coastal ecosystems, due to the range of climate change related pressures impacting them. These include: increased temperatures, changes in precipitation (duration, frequency, timing and intensity), alterations to storm frequency and intensity, changes in ocean currents and sea level rise (Ward et al., 2016a). These impacts combined with increased anthropogenic pressures, namely, pollution, land conversion, and resource extraction, means that coastal ecosystems are at risk from degradation and loss (Lacerda et al., 2021). This is particularly acute for vegetated coastal ecosystems such as mangroves, salt marshes and sea grasses (Ward et al., 2016b; Lima, 2020; Ward and Lacerda, 2021), and any alterations to these systems will have impacts on the communities that use or rely on the wide range of ecosystem services they provide (e.g., coastal protection and stabilisation, nursery habitat for commercially important fish species, tourism, and carbon sequestration, amongst others). Sea level rise has been noted as one of the most severe impacts to coastal ecosystems, particularly where coastal processes and soil and sediment supplies have been altered or disrupted (Ward et al., 2016c, Veettil et al., 2018). This can be particularly acute in semi-arid areas where fluvial sediment supply can be low, or urban coastal wetlands where natural processes are disrupted by municipal infrastructure (Ward and Lacerda 2021). However, these systems are fairly understudied in the literature, despite their global occurrence (North and South America, Middle East and Australia). Anthropogenic and climate change impacts can be further

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exacerbated where protections are limited or have been reduced opening the way for development.

In Brazil, where the Coastal Zone is a national heritage, as mandated by its Federal Constitution (Brasil 1988), coastal vegetation enjoys limited legal protection under Brazil's Forest Code (Brasil 2012). The country holds the largest extent of protected mangrove forests in the world, with 322,000 ha located within the boundaries of sustainable-use protected areas, e.g., *extractive reserves* (Resex) and *environmental protection areas* (APA) (Borges et al., 2017). However, it is argued that sustainable-use protected areas offer little *de facto* protection to coastal ecosystems (Prates et al., 2012). Furthermore, assessments of their overall effectiveness, legal enforcement and management performance are scarce. Mangroves are among the most impacted ecosystems along the Brazilian coast, despite their ecological, socio-economic and cultural importance (Ferreira and Lacerda, 2016; Lacerda et al., 2019).

Compounding this, 60% of Brazil's population live in highly urbanised areas located in low-elevation coastal zones (LECZs) that are naturally prone to coastal floods caused by the simultaneous occurrence of heavy rains and spring tides (Muehe and Neves, 1995; Muehe, 2010; Nicolodi and Petermann, 2010). Approximately 40% of Brazil's rural population live in LECZs, are poor and rely on natural barriers such as mangroves for protection against storms and other coastal hazards that are set to intensify in the near future (Barbier, 2015; Ward et al., 2016a). Because of rising sea levels, alterations to river dynamics, deforestation and land conversion, these natural barriers are eroding away and poverty-stricken coastal areas are becoming even more vulnerable to climate-related impacts (Lacerda et al., 2021). A report produced by the Brazilian Panel on Climate Change on climate change impacts, vulnerabilities and adaptation, concluded that several coastal areas in Brazil present high or very high vulnerability to climate change (PBMC, 2015). However, the lack of detailed assessments of Brazil's coastal vulnerability to the impacts of climate change, particularly with regards to sea level rise, prevents the identification of areas that are more at risk and the development of context-specific adaptation, contingency and mitigation strategies. In addition, these assessments consider the estuarine gradient as a whole, failing to account for the different hydro-geochemical gradients, which can trigger different responses from mangroves to a given driver, e.g. sea level rise or changes in fluvial inputs along the estuarine gradient.

In this study, we undertake an assessment of the vertical sediment accretion rate of urban mangroves in the semi-arid northeast of Brazil and how this compares with current local sea level rise trends, further considering factors that may have impacted their response to changes in hydrology and sediment availability, along the estuarine gradient.

2. Material and methods

2.1. Study area

The state of Ceará is located in the northeast of Brazil with a narrow coastal strip bordering a semiarid hinterland typified by a Bs climate (average annual rainfall 800 mm), and a detailed discussion of this section of coastline can be found in Ekau and Knoppers (1999) and Knoppers et al. (1999). The north eastern region of Brazil hosts ~69,000 ha of mangroves (~7% of the total mangroves of Brazil), typically tightly constrained to the immediate area of the low-lying coastal zone in the short estuarine reaches of the small rivers that discharge into the South Atlantic. The capital of the state of Ceará is Fortaleza, one of the largest urban areas in the northeast of Brazil, with a rapidly growing population estimated at ~2,703,000 in 2021 (IBGE, 2022). The rapid growth of this city is due to perceived opportunities for employment in the city that is well known as a tourist and transport hub in the region. This puts substantial and growing pressure on the coastal ecosystems as result of planned and illegal developments. This has been noted in the three urban mangrove environments in the city: Rio Ceará located the western edge of the urban area between Fortaleza and the municipal area of Caucaia, Rio Pacoti at the eastern edge of the city, and Rio Cocó that runs through the centre of the city (Fig. 1).

These three mangroves cover a total area of 1677 ha, (1158 ha Rio Ceará, 375 ha Rio Cocó, 144 ha Rio Pacoti [Lacerda et al., 2007]). These mangroves have a long history of use by the inhabitants for fisheries, salt production and crab/mollusc collection. The Rio Ceará mangrove forms part of the Rio Ceará APA, the Rio Cocó mangrove forms part of the State Park of Cocó and the Rio Pacoti mangrove forms part of the Rio Pacoti APA, all three administered by SEMACE (Ceará Superintendence for the Environment). The main mangrove species in these three estuaries are *Rhizophora mangle*, *Avicennia germinans*, *Avicennia shaueriana*, *Laguncularia racemosa*, and *Conocarpus erectus* (Lacerda et al., 2007). In all three sites the fringing mangroves where the cores were taken were dominated by a narrow band of *A. germinans* in the lower mangrove, backed by a wider band of *R. mangle* in the upper mangrove. In the Rio Pacoti and Rio Cocó sites the upper mangrove transitioned into saltmarsh vegetation on the landward side. The lower mangrove in the Rio Ceará site was the least densely vegetated, with fairly small, sparse *A. germinans*, as is typical in the lower mangrove in this site (Fig. 2). Ceará is characterised as mesotidal with tidal amplitudes ranging between 2 and 4 m) with the coastline typically consisting of sandy beaches, backed by dunes with occasional beach-rock outcrops interspersed with mangroves within the estuarine reaches (Godoy et al., 2018).

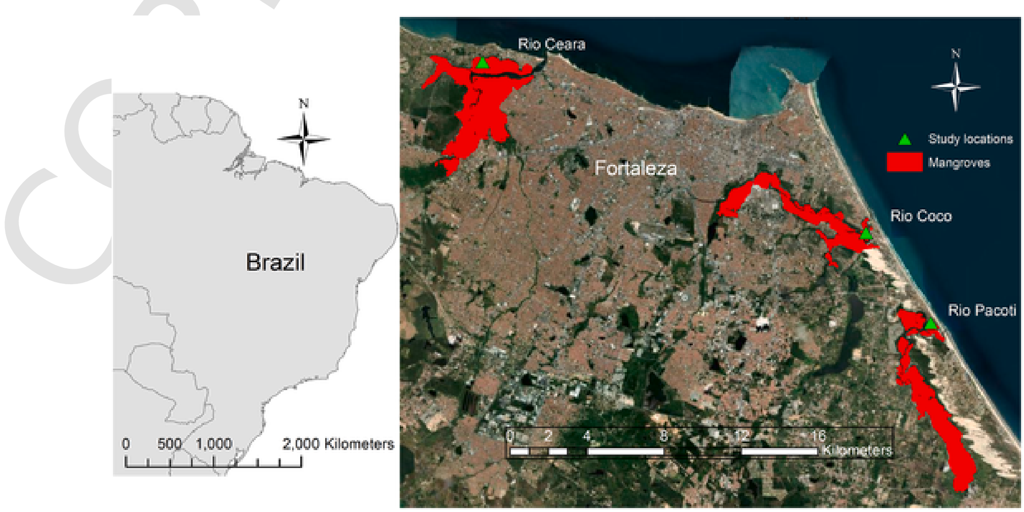

Fig. 1. Location of the study sites in the metropolitan area of Fortaleza in northeast Brazil.

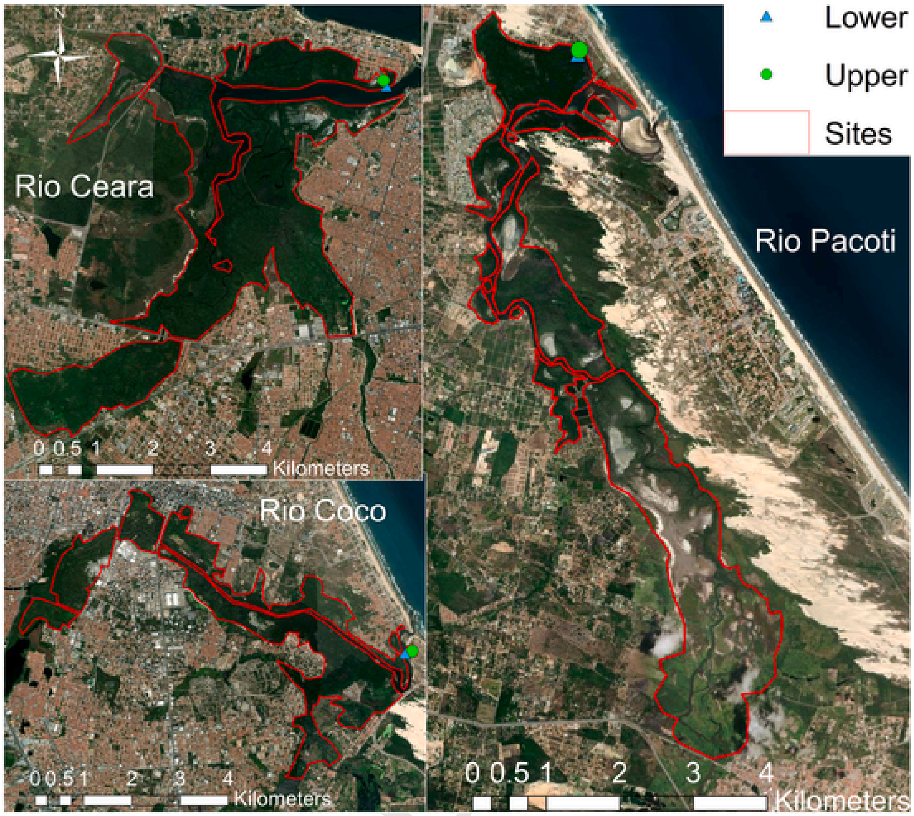

Fig. 2. Extent of the mangroves and apicum in all three study sites are bounded in red and the locations where the lower and upper cores were taken in the fringing mangroves are shown. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

2.2. Field methods

Samples were collected from the upper and lower fringing mangrove areas within each of the three estuaries, Rio Cocó, Rio Ceará and Rio Pacoti. The upper and lower mangroves are differentiated by species dominance (*R. mangle* in the upper and *A germinans* in the lower), and the difference in elevation is quite small. Upper mangrove samples were collected directly inland from the lower mangrove core and at least 15m from the edge of the boundary between the two. Two 10 cm diameter and 40 cm long PVC cores were driven into the soil to retrieve intact undisturbed depositional records. A walkover survey of the fringing mangroves was undertaken with a series of small (50 cm deep) trenches dug to evaluate the representativeness of the stratigraphy where the cores were taken (the stratigraphy was fairly homogeneous across the site) as per Ward et al. (2014). Care was taken to avoid areas with excessive bioturbation from burrowing crabs and to ensure that there was minimum (<10%) compaction of the sediment. Cores were sealed in the field and packed to ensure minimal impacts to the structure of the soil horizons and frozen until analysed in the laboratory (as per Ward, 2020a).

2.3. Sample preparation

The soils were removed from the core barrels by thawing the outer surface of the PVC tubes defrosting the outer edge of the sample while maintaining the majority of the core frozen and intact, enabling the soils to be removed from the barrels with no compaction. The cores were measured before and after removal to assess there was no compaction during the removal of the samples from the barrels. The outer edge of the soil from the cores was removed by scraping laterally to prevent down core mixing and contamination of the samples. The cores were then split into 1 cm depth sub-samples, oven dried at 40 °C and reweighed after 4, 5 and 6 days until they reached constant weight. The dry sub-samples were disaggregated and a part of the sample (~8 g) was weighed and placed into a cylindrical vial and sealed for 1 month prior to analysis in the gamma spectrometer (Ward, 2020a).

2.4. Loss-on-ignition and granulometry

The remaining part of the sub-samples were weighed in a crucible and then placed in a muffle furnace at 450 °C for 24 h (Lima, 2020). This was undertaken to ensure that there was full combustion of the organic material in the samples. Higher temperatures have been noted to produce overestimations of organic material through burning off clay-bound structural water, whereas lower temperatures may provide incomplete combustion of organic material. Ash contents as a result of loss on ignition can provide overestimations of fines, although loss on ignition can also result in underestimations of fines as a result strong binding of clays during heating (Vaasma, 2008). Following combustion, samples were reweighed to estimate organic matter content in the soils. Combusted samples were then placed in a Malvern Mastersizer 2000 laser particle size analyser and results were classified using the Wentworth (1922) scale. In order to prevent flocculation of clay particles, 10 ml of sodium hexametaphosphate was added to the samples which were then placed on a shaker for 30 min prior to particle size analysis. Within the particle size analyser samples underwent sonication and three separate runs of the sample were undertaken (standard error < 1% for all samples) as per Ward et al. (2014), Lima, (2020) and Ward (2020b).

Dry bulk density was evaluated using the methods and equations presented in Dadey et al. (1992) and further reported in Ward et al. (2014), and Ward (2020b).

2.5. ^{210}Pb and ^{137}Cs dating

Activity profiles of ^{210}Pb (half-life [t½] = 22.26 years) have been widely used to evaluate sedimentation rates in marine, coastal and lacustrine environments including coastal wetlands (Callaway et al., 1997; Teasdale et al., 2011; Ward et al., 2014; Arias-Ortiz et al., 2018). This is often undertaken using alpha or gamma spectroscopy, although using the latter both ^{210}Pb and ^{137}Cs levels can be determined and ^{137}Cs can be used to independently verify ^{210}Pb derived dating. In this study gamma spectroscopy was used and soil accretion rates were calculated using Constant Rate of Supply (CRS) model, which provides varying soil accretion rate values over time. The CRS model calculates specific ages for soil horizons using inventories. The total inventory in the core is determined using the sum of $^{210}Pb_{excess}$ × Dry bulk density × thickness of the core sub sample (in this case 1 cm). An overview of these methods can be found in Appleby and Oldfield (1992), Appleby (2001) and Cuesta et al. (2022).

Limits of gamma detection are variable dependent the gamma energy of the radionuclide, count time and mass of the sample (Teasdale et al., 2011). For this study, the limits were in the range of 2.8 Bq/kg for ^{210}Pb, and 0.18 Bq/kg for ^{137}Cs, for a 500,000 s count time and all error values were less than 5%.

In northeast Brazil, there is a clear ^{137}Cs marker signal from 1963 of the global input peaks of deposition that occurred prior to the signing of the weapons test ban treaty (Ritchie and McHenry, 1990; Foucher et al., 2021).

Atmospheric deposition of ^{210}Pb has not varied significantly based on the available records (Foucher et al., 2021) suggesting that the requirements of the CRS method have been met. ^{137}Cs atmospheric deposition has been reported as having a high spatial variability and there is a lack of data on this in the region, therefore it was not possible to utilise the ^{137}Cs:^{210}Pb inventory ration to assess erosion incidents (Plater and Appleby, 2004).

Gamma activity profiles were analysed using a Canberra ultra-low background high purity Germanium gamma spectrometer at the University of Brighton. The radionuclide spectra were accumulated using a 16k channel integrated multichannel analyser and analysed with the Genie™ 2000 system. Energy and efficiency calibrations were carried out using a bentonite clay standard spiked with a mixed gamma-emitting radionuclide standard, QCYK8163, and checked against an IAEA marine sediment certified reference material (IAEA 135) with background detection levels calibrated after each core run.

3. Results

3.1. Soil characteristics

Within the six cores from three sites core depths varied from 28 cm (Rio Ceará lower mangrove) to 37 cm (Rio Pacoti upper mangrove) (Fig. 3). In some sites (Rio Cocó and Rio Pacoti) there was a boundary layer of oysters at the bottom of the cores, in others there was simply some larger sediment, mixed with shells that resulted in losses at the bottom of the cores (e.g., Rio Ceará lower mangrove). Rio Ceará had the lowest percentage of organic matter, with little downcore variation (lower mangrove 1.7%–5.3%, upper mangrove 0.5%–3.5%), nor significant differences between the lower and upper mangroves (mean 1.7% and 3.1% respectively, $p < 0.05$) (Fig. 3, Table 1). In the lower and upper Rio Cocó cores, there was a decrease in organic matter from the surface to the bottom soils (over only 4 cm in the upper mangrove core) from ~13% in both cores to 3.7% and 2% in the lower and upper cores, respectively and no noted significant differences between the upper and lower mangroves ($p < 0.05$). A similar trend was noted in the Rio Pacoti lower mangrove core (from ~10% to 2.6% top to bottom), whereas there was little downcore variation in the Rio Pacoti Upper core although there were two strata with a trough and peak of 0.9%–7.1% respectively, with no significant differences between the lower and upper mangroves ($p < 0.05$).

In all the sites except Rio Cocó lower mangrove, the surface sediments were sand sized, which is not unexpected in this region (Fig. 4). The deeper sediments in the Rio Ceará lower mangrove and Rio Pacoti upper mangrove fluctuated between silt dominated and sand dominated (Fig. 4). All sites had low clay fractions typically between 0 (Rio Cocó upper mangrove) and 12% (Rio Pacoti upper mangrove), with one spike in clay to 19% toward the bottom of the Rio Cocó lower mangrove core (Fig. 4).

3.2. Radionuclide analysis

All cores showed a near exponential decline in ^{210}Pb activity from the surface to between 12 and 15 cm. In the Rio Ceará lower mangrove

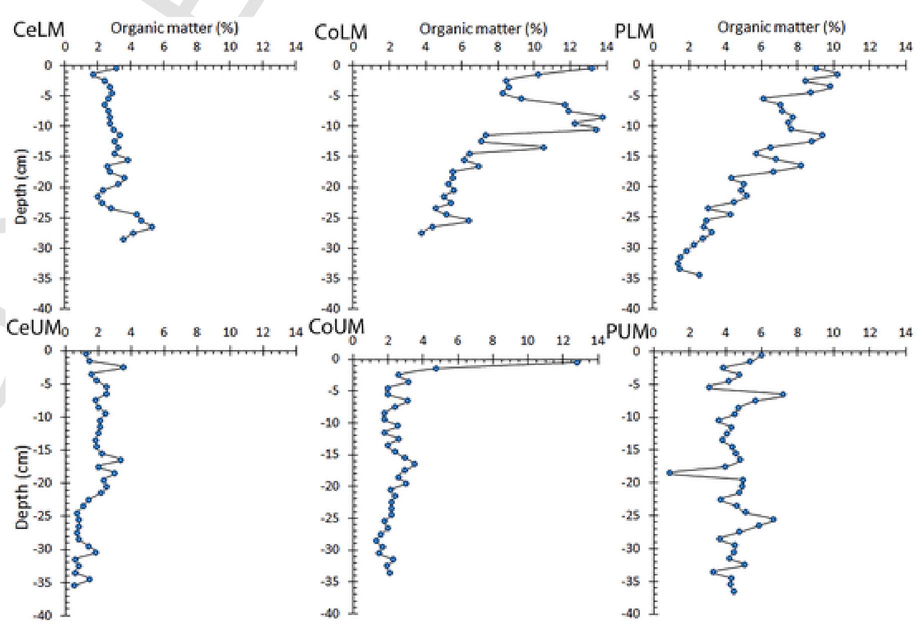

Fig. 3. Organic matter/depth profiles. CELM = Rio Ceará lower mangrove, CEUM = Rio Ceará upper mangrove, CoLM = Rio Cocó lower mangrove, CoUM = Rio Cocó upper mangrove, PLM = Rio Pacoti lower mangrove, and PUM = Rio Pacoti upper mangrove.

Table 1
Means and standard deviations of organic matter, granulometry and ^{214}Pb for all cores at all sites. LM is Lower mangrove and UM is upper mangrove.

Site	OM %	Clay %	Silt %	Sand %	214 Pb (Bq/kg
	Mean (SD)	Mean (SD)	Mean (SD)	Mean (SD)	Mean (SD)
Ceará LM	3.1 (0.8)	3.6 (1.9)	24.9 (11.7)	71.5 (13.6)	16.6 (1.5)
CearáUM	1.7 (0.8)	0.6 (0.6)	5.3 (3.0)	94.1 (3.6)	13.3 (0.9)
Cocó LM	7.0 (3.1)	7.0 (3.1)	69.4 (6.4)	23.6 (8.8)	22.9 (2.4)
Cocó UM	2.7 (1.9)	1.4 (1.0)	15.4 (7.6)	83.2 (8.4)	12.9 (1.0)
Pacoti LM	5.6 (2.7)	3.3 (1.9)	22.6 (7.9)	74.1 (9.7)	11.9 (2.7)
Pacoti UM	4.5 (1.0)	6.1 (3.2)	42.2 (18.5)	51.7 (21.7)	13.4 (1.1)

core there a substantial decrease in activity at 4 cm and 8 cm, which may have been related to bioturbation (often a problem in S. American mangrove soils due to crab burrowing). Similar limited disturbance can be noted in the Rio Cocó lower mangrove core (at 4–5 cm depth). Mean ^{214}Pb activity (used as a proxy for background ^{210}Pb) was between 11.9 Bq/kg and 12.9 Bq/kg dependent on site (Table 1, Fig. 5). This was comparable to the background readings of ^{210}Pb at the bottom of the cores for all sites (Fig. 5). There were some recorded fluctuations in surface ^{210}Pb activity in the core from Rio Ceará lower mangrove, and from both Rio Pacoti cores, potentially linked to deposition of older material at the surface, although levels were not substantially lower than in the other cores.

Activity levels of ^{137}Cs were fairly low for all sites, with a maximum activity of 0.71 Bq/kg, recorded in the Rio Ceará upper mangrove core (Fig. 6). There was limited evidence of post depositional relocation of ^{137}Cs in the cores, with the possible exception of the Rio Cocó lower mangrove core, where ^{137}Cs was recorded in a section dated 1992 using the ^{210}Pb CRS method.

Average rates of soil accretion derived from both the ^{137}Cs and ^{210}Pb methods show high agreement for all sites with the exception of the Rio Cocó lower mangrove site, which show substantially higher rates of sedimentation, particularly in the more recently deposited material (Table 2 and Fig. 7).

Dated soils using the ^{210}Pb CRS method reach back as far as the early 1900's (Rio Pacoti upper mangrove), although are typically around 85–90 years (1925–1930 for the Rio Cocó sites, the Rio Pacoti lower mangrove and the Rio Ceará lower mangrove) (Fig. 7).

Soil accretion rates in the Rio Ceará site vary between 1.7 and 2.2 mm/yr (Table 2), with higher rates of sedimentation recorded over time in the upper mangrove site. In the Rio Ceará lower mangrove there appears to be a peak in soil accretion in the late 1980's of 4.5 mm/yr with decreases in soil accretion in more recently deposited material (Fig. 7).

Both the lower and upper mangrove sites in the Rio Cocó estuary showed an increase in soil accretion in recent decades, and this is particularly acute in the lower mangrove site (10.5–15.5 mm/yr) (Fig. 7). Average rates of soil accretion are the highest in this estuary derived from both the ^{210}Pb and the ^{137}Cs results, but with different rates depending on method (Table 2).

Within the Rio Pacoti lower mangrove, soil accretion rates appear to have been increasing in recent decades, up to 4.1 mm/yr around 2002 and 3.4 mm/yr in the upper soil layer (Fig. 7). Soil accretion reached a peak (2.9 mm/yr) around 1989 in the Rio Pacoti upper mangrove although appears to be substantially lower in the most recently deposited soils (0.6 mm/yr) (Fig. 7).

As can be seen in Table 3, rates of local sea level rise in the mangroves are broadly in excess of soil accretion, with the exception of the Rio Cocó lower mangrove, where soil accretion rates have been very high over the last 20 years. In the Rio Cocó upper mangrove local sea level rise rates are only just in excess of soil accretion, and it appears that soil accretion is increasing, so these mangroves may not be under threat (Table 3 and Fig. 7). The mangroves with the lowest vertical accretion rates compared to sea level rise appear to be those within the Rio Pacoti upper mangrove (−2 mm/yr) and those in the Rio Ceará lower (−1.6 mm/yr) and upper (−1.3 mm/yr) (Table 3).

4. Discussion

4.1. Geochronology and soil accretion rates

The results of this study have shown that there is unlikely to have been significant post depositional remobilisation of ^{137}Cs, with the possible exception of the Rio Cocó lower mangrove core. The ^{137}Cs follows similar trends in the cores and there is a clear peak in activity, which in this area is likely to have only come from deposition linked to the above ground nuclear weapons test ban treaty (Foucher et al., 2021). Levels of this radionuclide are low across all sites, although this has been noted as typical for this region (Foucher et al., 2021). Peaks are fairly sudden,

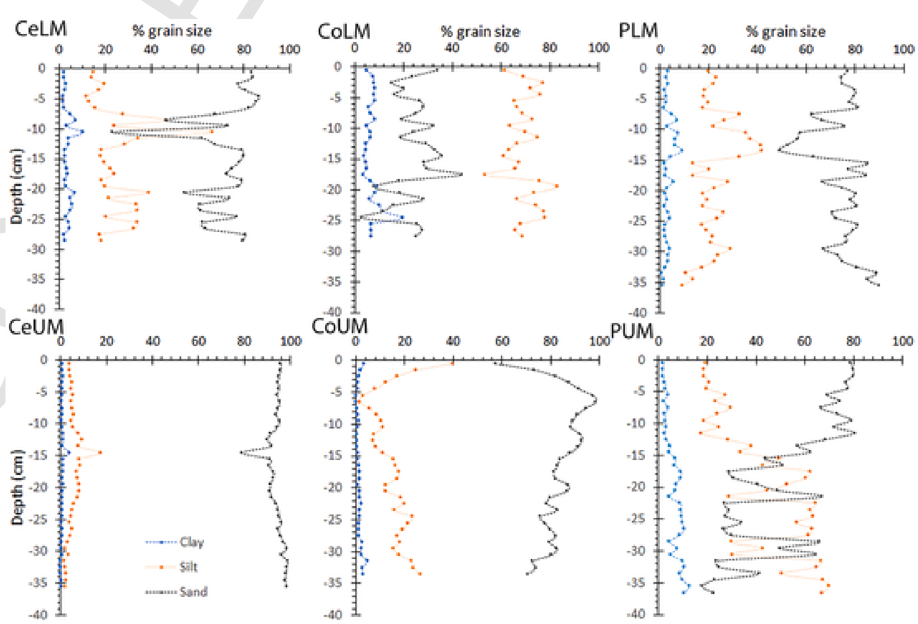

Fig. 4. Granulometry/depth profiles. CELM = Rio Ceará lower mangrove, CEUM = Rio Ceará upper mangrove, CoLM = Rio Cocó lower mangrove, CoUM = Rio Cocó upper mangrove, PLM = Rio Pacoti lower mangrove, and PUM = Rio Pacoti upper mangrove.

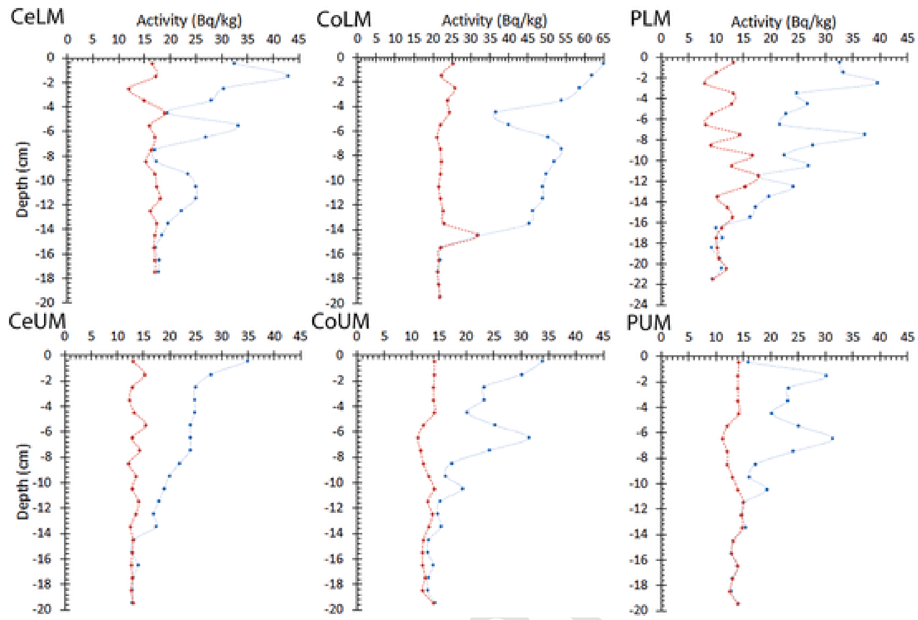

Fig. 5. Activity/depth profiles for $^{210}Pb_{excess}$ (blue circles) and ^{214}Pb (red diamonds) as a proxy for ^{210}Pb supported activity. CELM = Rio Ceará lower mangrove, CEUM = Rio Ceará upper mangrove, CoLM = Rio Cocó lower mangrove, CoUM = Rio Cocó upper mangrove, PLM = Rio Pacoti lower mangrove, and PUM = Rio Pacoti upper mangrove. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

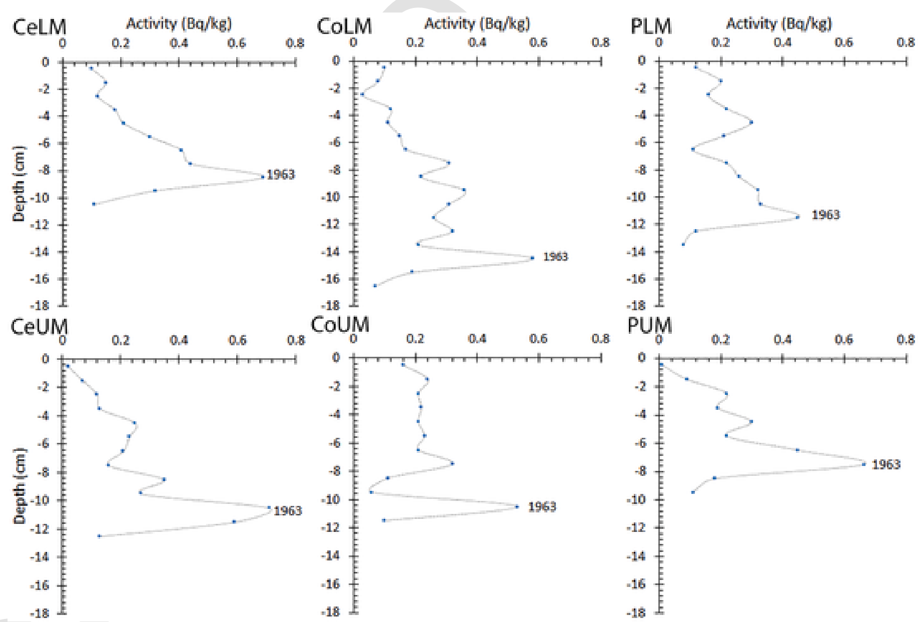

Fig. 6. ^{137}Cs activity with depth, showing the peak in deposition, which is likely to be derived from 1963 linked to the introduction of the above ground nuclear weapons test ban treaty. CELM = Rio Ceará lower mangrove, CEUM = Rio Ceará upper mangrove, CoLM = Rio Cocó lower mangrove, CoUM = Rio Cocó upper mangrove, PLM = Rio Pacoti lower mangrove, and PUM = Rio Pacoti upper mangrove.

with little broadening that would suggest remobilisation (Ward et al., 2014, 2020a). The stability of the ^{137}Cs is likely to be linked to the low organic matter (typically <10%, Fig. 3) in the soils, compared to many other coastal wetlands (Rosen et al., 2009). The sediments were a mix of sand and silt, although in this area it is predominantly fine sand that is recorded as a result of the strong influence of aeolian depositional processes in the region, which can be dominant in some areas (Ward and Lacerda 2021), particularly where fluvial discharge is low. Finer sediments are more likely to retain ^{210}Pb and ^{137}Cs in place, with little post depositional relocation (Walling and He, 1993; Cundy and Croudace, 1996; Borretzen and Salbu, 2002). Any disturbance in the soil matrix in the areas is likely to be from bioturbation, particularly by *Uca* spp. and the density of burrows and level of bioturbation is influenced by species type and their preference for specific soil types (Bezerra et al., 2006). While there were no evident traces of bioturbation in the cores, there may have been some post depositional relocation of ^{137}Cs in the Rio Cocó lower mangrove core to strata that predate the original introduction of the radionuclide as has been seen in other studies (Thomson et al., 2001; Teasdale et al., 2011). With the exception of the Rio Cocó lower mangrove core, the soil accretion rates derived from the ^{137}Cs impulse dating method are largely in agreement with those from the ^{210}Pb CRS method (Table 2). As has been shown in a large number of studies (Teasdale et al., 2011, Ward et al., 2014, Barsanti et al., 2020; Ward, 2020a, 2020b, Foucher et al., 2021), ^{137}Cs

Table 2
Average rates of soil accretion from the ^{210}Pb CRS dating method and ^{137}Cs methods for all sites. For comparison, the CRS derived average ^{210}Pb soil accretion rates since 1963 were provided. LM = lower mangrove and UM = upper mangrove. All soil accretion rates are in mm/yr.

Site	^{210}Pb CRS mean accretion rate	^{210}Pb CRS mean accretion rate since 1963	^{137}Cs accretion rate 1963
Ceará LM	1.7	1.9	1.7
CearáUM	2.0	2.2	2.1
Cocó LM	6.7	7.1	2.9
Cocó UM	2.1	3.1	2.1
Pacoti LM	2.2	2.6	2.3
Pacoti UM	1.2	1.5	1.5

can provide an independent verification of ^{210}Pb dating within coastal wetland environments, including the results presented in this study.

The levels of ^{210}Pb in this area are fairly low compared with many other global regions (Foucher et al., 2021). This is likely to be due to the low levels of precipitation in the region (<800 mm/yr), as a result of the semi-arid climate (Knoppers et al., 1999). ^{210}Pb deposition to surface soils is through precipitation, and where this is low it is likely that there will be a lower activity of this naturally occurring radionuclide (Mohan et al., 2019). However, as has been noted, the close similarity of ^{137}Cs and ^{210}Pb soil accretion rates in all sites suggests that the low levels of ^{210}Pb have not influenced the robustness of the dating analysis.

4.2. Changes in soil accretion over time

There are a wide range of factors that can influence soil accretion in coastal wetlands including precipitation, river damming, aquaculture, sea level change, tidal influences, available material, storm surges, aeolian processes and land-use/land cover change (Ward et al., 2016a). Of these, all but storm surges are likely to have a substantial influence on mangroves in northeast Brazil (Lacerda et al., 2020), as there are rarely very strong winds that occur in this region (Ward et al., 2016a). Where rates of sea level rise are in excess of soil accretion, there is likely to be a gradual loss of the mangrove extent at the seaward edges where it is not possible for landward migration. Sea level rise in the region is slightly higher (3.5 mm/yr) than global average rates (3.2 mm/yr Church and White, 2011). In the Rio Ceará and Rio Pacoti lower and upper mangroves, sea level rise is in excess of soil accretion (Table 3), meaning that there are likely to be losses at the littoral edge of the mangroves. The results from the Pacoti lower mangrove are however, within the uncertainty range. In the Rio Ceará mangroves, the landward edge is partly bounded by illegal urban settlements, meaning that landward migration is no longer possible. There are some former mangrove areas in Rio Ceará occupied by abandoned shrimp aquaculture ponds where recolonisation by mangroves is possible. However, in many areas there have been noted issues with recolonisation of former shrimp ponds by mangroves, including inappropriate hydrology and hydrological connectivity (Matsui et al., 2010), high levels of contaminants (Lacerda et al., 2020), and unsuitable geomorphology for mangrove survival (Matsui et al., 2010). However, to the west of the estuary there is a large low-lying area covered by a mixture of salt marsh, apicum and terrestrial vegetation that could be occupied by the landward migration of mangroves. These losses, particularly of the apicum and salt marshes mean that the distinctive ecosystem services provided by these habitats (e.g., recreation, grazing, supporting distinct fauna and flora, harvesting crustaceans) would be lost (Albuquerque et al., 2014). The Rio Ceará site was also the only site to have lower rates of soil accretion in the lower mangrove than the upper mangrove, although this was most likely as a result of the less dense vegetation cover of the mangroves in this zone.

In the Rio Pacoti sites, vertical accretion compared to sea level rise in the upper mangrove is greatest (Table 3), which may result in alteration in vegetation cover (migration of lower mangrove into the upper mangrove area) or increased stress to the plants in the upper mangrove system through more frequent inundation. As has been noted for the Rio Ceará mangroves, large sections of the Rio Pacoti mangrove are bounded by urban infrastructure (houses, roads, etc.), as well as some small aquaculture ponds, predominantly to the west (in the direction of the city Fig. 1). To the east, the majority of the estuary is bounded by

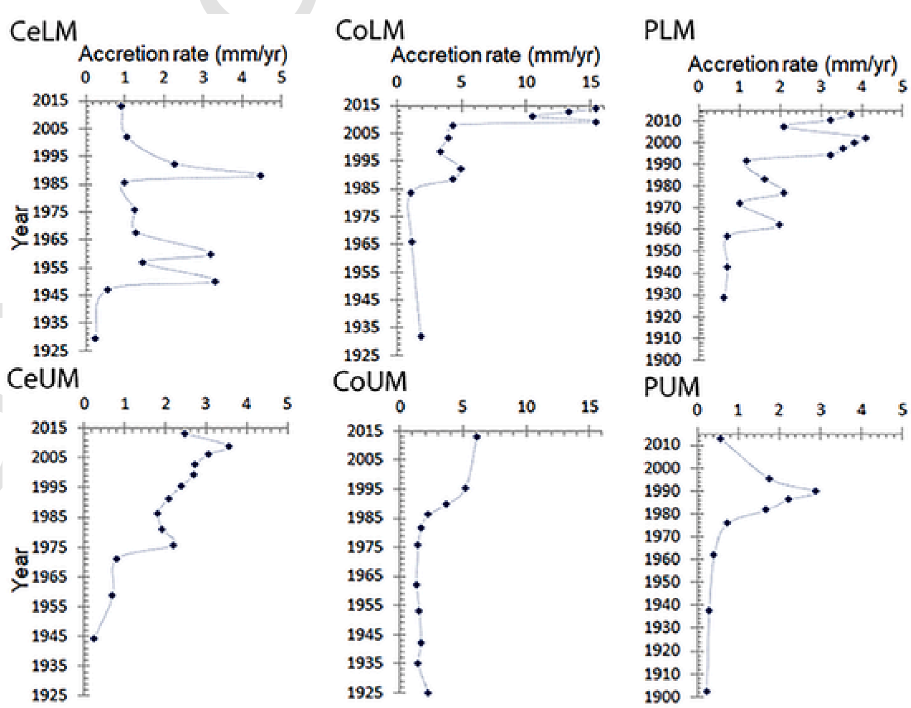

Fig. 7. Sedimentation rates over time derived from the CRS ^{210}Pb dating analysis. CELM = Rio Ceará lower mangrove, CEUM = Rio Ceará upper mangrove, CoLM = Rio Cocó lower mangrove, CoUM = Rio Cocó upper mangrove, PLM = Rio Pacoti lower mangrove, and PUM = Rio Pacoti upper mangrove.

Table 3
Mean accretion rates for all sites compared with recorded sea level rise (from the tide gauge station at the Port of Mucuripe in the city of Fortaleza (12 km from Rio Ceará, 8 km from Rio Cocó and 14 km from Rio Pacoti, the gauge has been in place since 1948). All rates are in mm/yr (standard deviations of CRS accretion rates are in brackets).

Sites	^{210}Pb CRS mean accretion rate since 1963	Sea level rise	Local sea level rise
Ceará LM	1.9 (1.3)	3.5	−1.6
Ceará UM	2.2 (0.7))	3.5	−1.3
Cocó LM	7.1 (5.5)	3.5	3.6
Cocó UM	3.1 (1.9)	3.5	−0.4
Pacoti LM	2.6 (1.1)	3.5	−0.9
Pacoti UM	1.5 (1)	3.5	−2

dunes, with the dominant wind direction moving the dunes into the estuary. This means that during periods of low flow, dune encroachment into the estuary and mangroves can be high and mangrove migration into these areas is unlikely (Ward and Lacerda, 2021). Within River Pacoti the greatest increase in mangrove cover observed by Godoy (2015) was due to the recolonisation of abandoned salt ponds and enlarged fluvial beaches resulting from damming of the river upstream (Lacerda et al., 2007).

The mangroves in Rio Cocó are tightly bounded by urban infrastructure on both sides and there has been substantial urban encroachment into the mangrove over the last few decades including the building of a large shopping centre (Shopping Iguatemi), completed in 1982 and with further losses to the mangrove in 2008, and again in 2011–2015. The exclusively urban location of the Rio Cocó mangroves means that there is nowhere for the ecosystem to migrate to. However, this mangrove is the least threatened of the sites studied, as in the last three decades soil accretion has exceeded sea level rise in the both the lower and upper mangrove (Fig. 7). Soil accretion, particularly in recent decades has substantially increased, yet fluvial influences have not substantially altered over this time period, and connectivity with dunes and their sediment supply has substantially decreased over time due to the increase in municipal infrastructure, predominantly built directly on the dunes. The city's storm drainage however, directly feeds into Rio Cocó and this may be providing a substantial amount of material to the estuary, and subsequently to the mangroves, allowing them to keep pace with sea level rise. This, however, may have negative impacts on ecosystem health due to potential pollution inputs from urban sources (waste water and urban runoff). These anthropogenic sources emit over 2400 t yr^{-1} and 600 t yr^{-1} of nitrogen and phosphorus, respectively, to the Cocó River estuary (Lacerda et al., 2008). This has been seen in other urban coastal wetlands globally (Ahmed et al., 2018, Abdelhady et al., 2019, Cundy and Croudace, 1996; Celis Hernandez et al. 2020a, 2020b, 2021a, 2021b, 2022). However, nutrient inputs have been shown to accelerate growth in mangroves in North America and Australia, particularly enhancing the growth of shoots (Lovelock et al., 2004, 2009). This has been suggested to be driven by their high plasticity in response to nutrient availability, as they are adapted to low nutrient environments, but can exploit high nutrient levels when available (Reef et al., 2010). However, in spite of this plasticity, it has been suggested that long-term exposure to high nutrient levels, particularly in sites with low rainfall inputs and high soil salinities, such as occur in the mangroves of north east Brazil particularly in the fringing mangroves, can result in enhanced tree mortality (Lovelock et al., 2009).

All three sites, witnessed substantial increases in estuarine mangrove extent between 1992 and 2011 (Godoy, 2015), with increases varying from 22.8% in Rio Cocó, 28,3% in Rio Ceará to a maximum of 47.8% in Rio Pacoti (Table 4). Recolonisation of abandoned salt ponds (Cocó and Pacoti) and aquaculture ponds (Ceará) were the major drivers of this expansion. In addition, increased sedimentation areas along river margins and islands have contributed to this increase in area due

Table 4
Change in mangrove area cover between 1992 and 2011 as evaluated using Landsat imagery (adapted from Godoy, 2015).

Mangrove site	1992 mangrove area (ha)	2011 mangrove area (ha)	Change in area (ha)	% change in area
Rio Pacoti	230	340	+110	+47.8
Rio Cocó	368	452	+84	+22.8
Rio Ceara	566	726	+160	+28.3

to decreased sediment export to the mouth and coast, due to reduced water flow as a result of river damming (Lacerda et al., 2007). More recently, active replanting in the Pacoti and Cocó rivers may contribute to additional increases although this was not recorded in 2011. Unfortunately, however, available area for mangrove migration is restricted in all three sites due to urban development expanding at the edge of estuaries. Also, large dune systems, typical of this coastline are natural barriers to mangrove expansion (Lacerda, 2018) and may threaten them as their mobility is accelerated by global climate change (Maia et al., 2005).

In the fringing mangroves sediment limitation is suggested as one of the key factors that has influenced vertical accretion of these three sites including alterations to marine and fluvial sediment transport. In the case of the Pacoti River, the Redenção dam, controls river flow at about 1 m^3 per second. Previously, in the rainy season flow varied from 0 to 19 m^3 per second. The high flows of the rainy season disappeared, resulting in trapping sediments within the upper estuary (Lacerda et al., 2007).

The Ceará and Cocó rivers were also dammed, but the flows before and after installation were not calculated, although anecdotal evidence suggests that the decrease in discharge has been notable in the rainy season.

In addition to controls on river flows, there has also been a reduction in annual rainfall ranged from 4.8 to 5.6 mm year^{-1} over the last 30 years throughout the state, reducing river flows and increasing sediment retention in estuaries (Moncunill, 2006; Alvalá et al., 2019; Marengo et al., 2018).

The Rio Ceará Estuary is the only site down coast of the large jetties of Praia de Iracema and the port of Mucuripe, the mouth of the river is therefore less likely to be supplied by sediment from marine transport processes and the coast is more subject to erosion than the Rio Cocó and Pacoti estuaries that are located up coast of the jetties (Godoy et al., 2018; Targino da Silva et al., 2019).

It should be noted that soil accretion is not the only factor that contributes to surface elevation change, and several authors have pointed out that organic matter input (Krauss et al., 2014; Ward et al., 2016a; Schuerch et al., 2018), and root growth may influence surface elevation change and either allow coastal wetlands to keep pace with sea level rise (Ward, 2020b) or exacerbate the issue, particularly where degradation or stress lead to decreases in productivity or root death/decomposition (Mafi Gholami et al., 2018; Celis Hernandez et al., 2022). The high inputs of organic material, particularly in the Rio Cocó mangroves and the Rio Pacoti lower mangrove may partially offset sea level rise, in combination with minerogenic inputs.

4.3. Future of urban mangroves in northeast Brazil

In Northeast Brazil, the coastal zone borders the semi-arid inlands with limited geomorphological obstruction to mangrove migration and landward expansion has been recorded in the order of 20–30% in some areas, including the three studied here (Maia et al., 2006; Godoy, 2015). While sea level rise is not particularly high compared to some global regions, local drivers can intensify the effects. In some sites in northeast Brazil, sea level rise combined with decreases in precipitation has been shown to have driven large-scale expansion of estuarine mangroves (e.g., Jaguaribe River in Ceará, Brazil) (Godoy and Lacerda,

2014) due to landward migration and increased growth of estuarine bars.

Whilst this response may be positive, other studies in the region have shown that the residence time of trace element contaminants substantially increased due to strengthening ocean forcing on the continental shelf, because of excess heat accumulating in the South Atlantic (a substantial issue as has been noted in various global regions in 2022 (Jacox et al., 2022), linked to climate change (Dias et al., 2013; Lacerda et al., 2013, 2020).

In the study in the Jaguaribe River, Godoy and Lacerda (2014) noted that while overall expansion of the mangroves was recorded, in the same area fringing mangroves, such as those studied in this research, were being eroded. They also noted, as with this study, that where progradation is unlikely to take place (e.g., as a result of low sedimentation rates) migration inland may be limited by dune encroachment, as has been seen in several areas of northeast Brazil (Lacerda, 2018; Ward and Lacerda, 2021) and is also likely in the Rio Pacoti mangroves in this study.

While mangroves are protected by legislation in most countries where they occur, legislation is often weak or non-existent regarding *apicuns* (hypersaline salt flats, in some regions known as *sabkhas*) (Lokier, 2013). In Brazil, one of the areas with the largest cover of mangroves in the world, the principal legislation protecting mangroves is the Forest Code created 1934 and updated in 1965 and 2012. This not only covers the protection of mangroves but also *apicuns* as part of the mangrove ecosystem continuum, however, the recent view by planners that these were permanently dry areas, led to a change in the latest version of the Forest Code, opening these up to development (Ward and Lacerda, 2021). In the northeast of Brazil alone, this revision opened an area of 600,000 ha to development that could have the potential to provide a buffer zone for landward migration of mangroves, including Rio Ceará, one of the sites in this study. This alteration to Brazilian legislation could drastically reduce the resilience of mangroves in northeast Brazil to sea level rise (Lacerda et al., 2022).

While there is a lack of consensus regarding how mangrove forests in northeast Brazil will respond to changes in rates of sea-level rise, inland migration seems to be a regional response where mangroves are free to migrate (Lacerda et al., 2019). This study demonstrates that further landward migration may not be possible in the study sites. The existing multilevel legal protection mechanisms in place have not ensured the protection of these mangroves against many anthropogenic pressures, consequently affecting their resilience to sea-level rise (Gilman et al., 2008).

Allowing mangrove forests to recover and restoring areas where they have been degraded are of paramount importance for the maintenance of the ecosystem services supplied by them (e.g., nursery habitats for fish, fishery catches and biodiversity and aesthetic appreciation for tourism). These, in turn, can support livelihoods and cultural values of coastal communities that depend directly on them (Lacerda et al., 2019). While the fate of urban mangrove forests hangs in the balance, so does the fates of the ecosystem services and associated socioeconomic benefits they provide.

5. Conclusions

The results of this study suggest that with limitations to areas for landward migration, semi-arid fringing urban mangroves may be more at threat from sea level rise than other mangroves at the seaward edge. This can be exacerbated by infrastructure alterations to sediment supply, which could vary from river damming to aeolian sediment sources being limited by dune fixation or simply building on top of dunes. In the sites studied here, poor or no urban planning may be partially to blame, particularly where illegal settlements encroach into mangroves. In semiarid areas where limitations to precipitation are already occurring and mangroves are in a 'stressed' environment (high salinity, irregular inundation regimes), decreases in sediment supply with resultant low vertical accretion rates may exacerbate responses to sea level rise and lead to mangrove degradation or loss at the seaward edge. However, where drainage from large urban conurbations drains directly into mangrove areas or adjacent estuaries, the sediment run off from urban sources may partially compensate for other alterations to available sediment. It should be noted that other researchers however, have found that urban mangroves can be stressed as a result of trace element contamination most likely derived from sediment run off from urban storm drainage systems.

Recent changes to Brazilian legislation may mean that landward migration into apicum areas is limited due to the opening up of development opportunities. Landward migration is also likely to be limited in predominantly urban mangroves and those that are bounded to the west (in this region) by dunes as a result of the dominant aeolian influences, which are also increasing as a result of climate change. Furthermore, while landward migration may be possible in some areas, this is likely to lead to the losses of lower fringing mangroves and their associated ecosystem services as has been noted by other researchers.

Brazil is among the 15 nations that are most-at-risk of climate change related impacts (Barbier, 2015). Strengthening the protection of a natural asset that is highly productive and offers a natural barrier against some of the hazards associated with climate change (e.g., mangrove forest), seems to be a logical way to ensure the provision of mangrove-derived benefits and adapt to the climate-related impacts well underway.

CRediT authorship contribution statement

Raymond D. Ward : Writing – review & editing, Writing – original draft, Validation, Resources, Project administration, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Luiz Drude de Lacerda :** Writing – review & editing. **Aline da Silva Cerqueira :** Writing – review & editing. **Vitor Hugo :** Writing – review & editing. **Omar Celis Hernandez :** Writing – review & editing.

Declaration of competing interest

The authors state they have no conflicts of interest in this submission.

Data availability

Data will be made available on request.

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