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Accelerated migration of mangroves indicate large-scale saltwater intrusion in Amazon coastal wetlands

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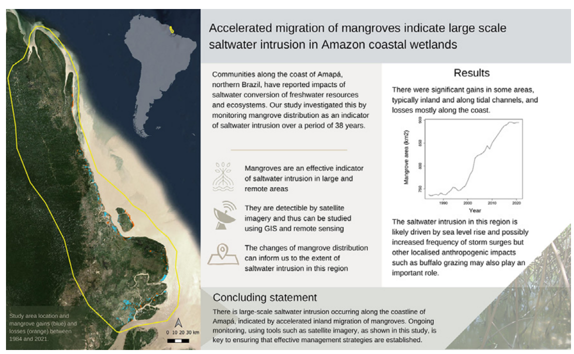
HIGHLIGHTS

- Saltwater intrusion in freshwater habitats impacts the delivery of ecosystem services.
- Saltwater intrusion is challenging to monitor in remote parts of the world.
- Time-series satellite imagery of coastal vegetation changes reveals saltwater intrusion.
- Mangroves expanded inland along the northern coast of Brazil over the last 38 years.
- Monitoring can help design actions to reduce impacts on biodiversity and livelihoods of coastal communities.

GRAPHICAL ABSTRACT



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ABSTRACT

Saltwater intrusion can dramatically transform coastal ecosystems, changing vegetation and impacting wildlife and human communities who rely on these natural resources. This phenomenon is difficult to measure over large and remote areas but can be inferred from changes in the distribution of salt-tolerant vegetation, such as mangroves, observable from satellite imagery. The northern coast of Brazil has the largest continuous mangrove forest in the world and very low human occupation. Even so, saltwater intrusion and changes to the coastline have been reported in this region, with potential consequences for mangrove carbon storage and for local livelihoods, but this has not been quantified due to the remoteness of the area. This study measured changes in mangrove distribution along the Northern Brazil coast in the state of Amapá, covering ca. 15,000 km^2, over the last 38 years using Landsat satellite imagery. We found that mangrove area in this region is highly dynamic, with significant gains and losses occurring over the study period, but with an overall net gain of 157 km^2. Mangroves have been systematically expanding inland and this growth has accelerated close to the shoreline and at the head of tidal channels in the last two decades, indicating rapid and large-scale saltwater intrusion in this region. This phenomenon is likely driven by sea level rise, which also accelerated in this region in recent decades, but anthropogenic impacts such as buffalo grazing may also play an important role.

1. Introduction

Saltwater intrusion is among the most direct and dramatic consequences of climate change (Wong et al., 2013). This phenomenon is largely driven by sea level rise (Bhattachan et al., 2019; Bowman et al., 2010;

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Minick et al., 2019) and magnified by coastal erosion and storm events (Wong et al., 2013). Saltwater intrusion may severely transform coastal ecosystems (Knighton et al., 1991; Minick et al., 2019). The saltwater contamination of soil and freshwater wetlands leads to biogeochemical changes (Herbert et al., 2015; Tully et al., 2019), altered inorganic chemical interactions, and disruption of important nutrient cycles (Herbert et al., 2015). These changes in the soil have flow-on effects on ecosystem function, with impacts on species interactions, distribution, and survival (Herbert et al., 2015; Neubauer et al., 2013). In particular, they can lead to the transformation of ecosystems through vegetation shifts, promoting the colonisation of salt-tolerant vegetation and the loss of salt-sensitive species (Kaplan et al., 2010; Tully et al., 2019). These shifts impact the structure of biological assemblages, resulting in the loss of species and ecosystems that support agriculture and fishery industries (Bhattachan et al., 2019; Castello and Macedo, 2016; Rahman et al., 2018), ultimately affecting the livelihoods of coastal human populations (Bhattachan et al., 2019; Rahman et al., 2018). These large-scale changes resulting from saltwater intrusion are occurring in coastal regions across the globe (Barbarella et al., 2015; Mahmuduzzaman et al., 2014; Mulrennan and Woodroffe, 1998; Wan et al., 2015).

There are many challenges in detecting and measuring saltwater intrusion as it is a largely invisible phenomenon (Rahman et al., 2018). Saltwater intrusion monitoring often involves on-the-ground measurement of groundwater salinity (Rahman et al., 2018), geochemical and physical characterisation (Rachid et al., 2017), and the use of sophisticated technology such as geoelectromagnetic techniques (Melloul and Goldenberg, 1997). These methods are expensive and time-consuming and therefore not suitable for covering large and remote areas (Barbarella et al., 2015; Bell et al., 2001; Rahman et al., 2018). An alternative monitoring approach is measuring visible ecological changes using remote sensing and GIS techniques (Barbarella et al., 2015; Rahman et al., 2018; Younes Cárdenas et al., 2017). Since salinity is a major limiting factor to plant growth, its effects can be observed at the whole plant level (Barbarella et al., 2015). The distribution of salt-tolerant vegetation can therefore provide a spatial overview of salinity distribution (Barbarella et al., 2015).

Mangrove forests are key features of coastal areas in tropical and subtropical regions, and they have distinct distributions and predictable responses to environmental factors, such as coastal erosion and accretion (Gilman et al., 2008; Nascimento et al., 2013; Souza Filho et al., 2006; Tran Thi et al., 2014). Mangrove distribution can provide important insight into saltwater intrusion, as the progress of salinity further inland favours the establishment of new mangrove (Barik et al., 2018). Mangrove forests can be easily identified from satellite imagery (Grieger et al., 2019; Younes Cárdenas et al., 2017), as they have distinct textures, colours, shapes, and positions in the landscape (Moity et al., 2019; Nascimento et al., 2013). For this reason, mangrove distributional changes determined from time-series satellite imagery have been used to infer on saltwater intrusion (Asbridge et al., 2016; Di Nitto et al., 2014; Liu et al., 2019; Souza Filho et al., 2006).

Brazil has the largest continuous mangrove forest in the world, extending along the eastern border of the Brazilian Amazon across the states of Maranhão, Pará, and Amapá (Nascimento et al., 2013). This region has high biodiversity, with large populations of waterbirds, unique wetland habitats, and several threatened vertebrate species such as the Amazonian and the West Indian Manatees (*Trichechus inunguis* and *Trichechus manatus*) (Domning, 1981). In addition, this region holds carbon stored in peat deposits that are among the largest in the world (Gumbricht et al., 2017). However, recent evidence and reports from fishing communities in Amapá have suggested that saltwater intrusion may be affecting large coastal areas in this region (Silva dos Santos et al., 2018). High salinity measurements were recorded in 2016 in a large freshwater lake (da Cunha and Sternberg, 2018; Silva dos Santos et al., 2018), and a smaller freshwater lake has become saline over the last decade. Given the rapid sea level rise observed by tide gauges in this region (Holgate et al., 2013) and the low topography of coastal areas (Anthony et al., 2010), we hypothesized that large-scale saltwater intrusion occurring in this region would be reflected by a gradual inland shift of mangroves over the last few decades. To assess the severity and scale of this phenomenon, we mapped and quantified changes in mangrove distribution along the coast of Amapá over the last 38 years using Landsat satellite imagery.

2. Methods

2.1. Study area

The study area encompasses ca. 520 km of coastline across the state of Amapá extending inland 25–35 km and including the islands in proximity to the coast (Fig. 1). Overall, it covers ca. 15,000 km^2 being equivalent to the size of Northern Ireland. Amapá is the northernmost state of Brazil and lies across the equator, thereby the climate is warm and humid with temperatures between 22 and 32 °C throughout the year and the wet season is between December and July with an average annual rainfall of 1000 mm. The Brazilian Amazon Rainforest covers a large proportion of the Amapá state (Ferreira and Lacerda, 2016), however the coastal zone consists of freshwater wetlands, saltmarshes, mangrove forests, and mudflats (Ferreira and Lacerda, 2016; Maria et al., 2017).

The main economic activities are derived from livestock farming and fisheries (Gomes et al., 2011). Water buffalo farming is a particularly successful industry as buffalo are highly adapted to the floodplain environment (Camarao et al., 2004; Sheikh et al., 2006).

The Amapá coastline is strongly influenced by sediment redistribution (Maria et al., 2017) and freshwater discharge of the Amazon River delta (Ferreira and Lacerda, 2016), which is located in the south of the state. Tidal forces, river discharge, wind, and currents all contribute to highly dynamic processes of erosion and accretion, channelisation, flooding, and salinisation in the study area (Silva dos Santos et al., 2018).

Mangrove forests in the study area comprise largely of black mangrove (*Avicennia germinans* and *schaueriana*), red mangrove (*Rhizophora mangle*) and white mangrove (*Laguncularia racemosa*) (Schaeffer-Novelli et al., 1990). Black mangrove is the dominant species in areas of higher elevation, while red mangrove is the most common species in areas of stronger tidal influence, such as the edge of tidal channels, river estuaries and offshore islands (Schaeffer-Novelli et al., 1990). White mangrove is also common in the study area, but mangrove forest dominated by these species was never observed during the fieldwork. During an aerial survey along the coastline, we observed that mangrove forest was well developed (ca. 15–20 m high) in most of the middle section of the study area, from the mouth of River Cassiporé (3.907° N, −51.158° E) to the mouth of River Sucuriju (1.674° N, −49.923° E), including the island of Maracá (2.019° N, −50.430° E). This section of the study area also shows signs of severe coastal erosion. In contrast, the northernmost section of the study area, around Cabo Orange (4.445° N, −51.537° E), and the section towards the south of the mouth of River Sucuriju, included large areas of accretion with patches of young mangrove forest (less than 5 m high).

Evidence of saltwater intrusion has been reported in two coastal freshwater lakes. Increased salinity was observed in the Piratuba Lake (1.684° N, −50.138° E) in April 2016 (da Cunha and Sternberg, 2018) and a severe transformation of Bom Nome lake (1.949° N, −50.624° E) over the last decade was reported by local communities, with the replacement of freshwater vegetation and fish by saline tolerant species.

2.2. Extracting mangrove distribution data from Landsat imagery

Landsat images (sensors 5, 7, and 8) were collated (sourced from USGS www.usgs.gov) over 38 years between 1984 and 2021. Three images were selected per year, filtered for dry season (August–November), low cloud cover, and processing level (preferentially L1TP (Precision and Terrain Correction) or L1GT (Systematic Terrain Correction)). The use of multiple images per year was to account for gaps due to cloud cover and Landsat 7 scan line corrector errors, and poor contrast of mangrove patches in some images. In some years (1984–1986, 1988–1989, 1992–1997, 1999) the selection criteria had to be relaxed because of a shortage of images available.

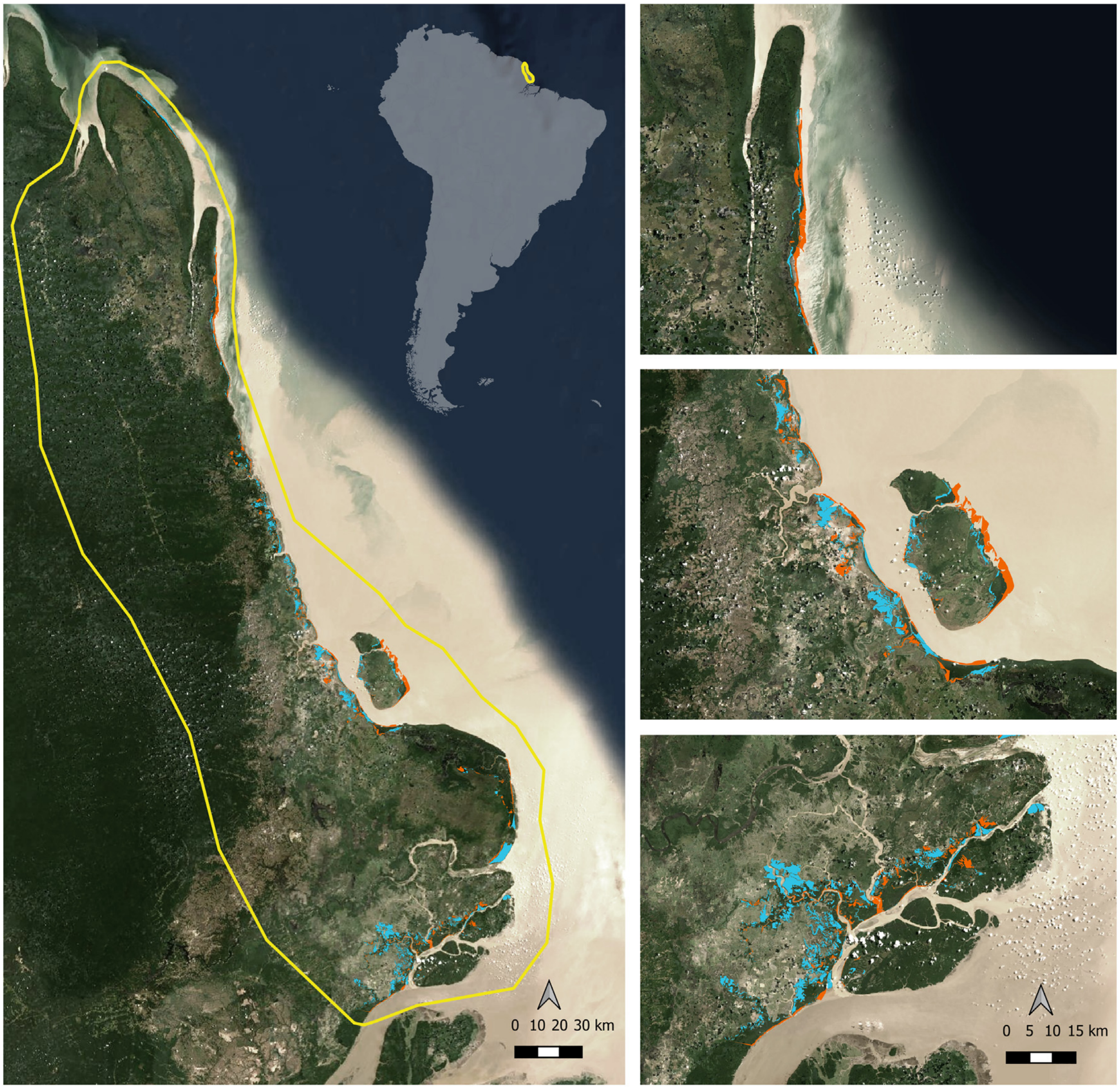

Fig. 1. Study area location and mangrove gains (blue) and losses (orange) between 1984 and 2021. Only mangrove patches with distinct boundaries with the surrounding environment were classified (see Methods for details). Background image – ESRI Satellite (ArcGIS/World_Imagery).

When L1TP or L1GT images were not available, L1GS (Systematic Correction) images (33 cases over 13 years) were used, and these required geometric correction. A geometric correction was performed in QGIS (QGIS Development Team, 2022) with the Georeferencer GDAL Plugin using Thin Plate Spline transformation and Nearest Neighbour resampling. These images were compared against georeferenced images (L1TP or L1GT) using ground control points based on predefined landmarks. The images were clipped to the study area prior to correction. For optimal contrast of mangrove patches, Red-Green-Blue spectral bands (30 m resolution) were selected for true colour composites, and brightness, contrast, and saturation were enhanced in QGIS.

Mangrove patches were identified from Landsat images based on colour (medium-dark green), texture (cauliflower pattern), shape (dendritic perimeter), and location (close to the coastline) (Moity et al., 2019; Nascimento et al., 2013). Patches with distinct boundaries to the surrounding environment were selected, typically bordered by freshwater marshes inland and by the sea or the edge of large channels extending from the coast. Polygons were drawn around mangrove patches in QGIS using Landsat images from 2021 and were then validated using high resolution satellite imagery (sourced from TerraMetrics www.terrametrics.com and Maxar technologies www.maxar.com) and aerial photography as a reference (acquired during an aerial survey at ca. 50 m high along the coastline). Each polygon was then duplicated and adjusted for the preceding year. This step was repeated for each consecutive year between 2021 and 1984.

For modelling purposes, polygons representing mangrove patches were intercepted with a 500 × 500 m grid. This resolution is fine enough to reflect short-term changes in the limits of mangrove patches but not too fine as to add redundancy to our models. Cells that had no mangroves over the 38 years or that overlapped with areas of coastline erosion and accretion were excluded. Each grid cell was classified as 1 or 0 if mangrove was present or not in each year.

2.3. Data analysis

Mangrove distribution changes across coastal and inland areas were modelled over time with a binomial Generalised Additive Mixed Model (GAMM). GAMMs are a non-parametric alternative to linear models, which allow the modelling of complex non-linear relationships between variables (Wood, 2017). GAMMs were used because mangrove growth is known to depend on factors that may present non-linear temporal trends, such as sea level rise (Boon and Mitchell, 2015) and deforestation (Leadley et al., 2014). Mangrove occurrence in grid cells was included as a binomial response variable in the model (0 – absence or 1 – presence), and sampling year was incorporated as a smooth term interacting with

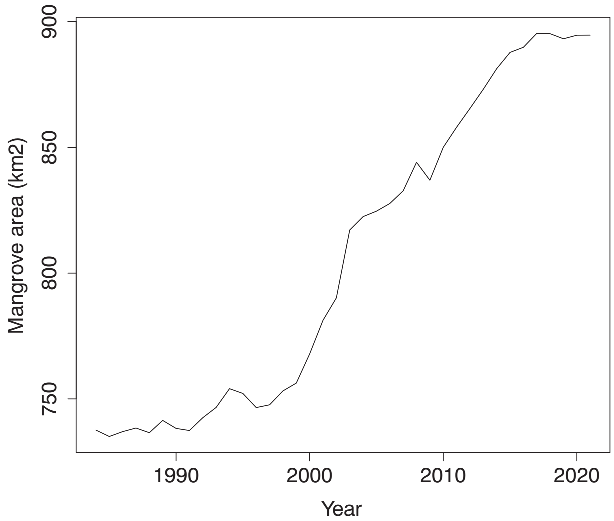

Fig. 2. Overall increasing trend of mangrove patches surveyed between 1984 and 2021.

distance from the coastline, which was categorized into three classes: (1) Coastal (up to 3 km to the coastline), (2) Mid (3–9 km to the coastline), and (3) Inland (further than 9 km to the coastline). The limits of distance classes were chosen in such a way that classes had a similar number of cells. Cell ID was also included as a random factor to account for the repeated measures of the same cells each year. Spatial autocorrelation was accounted for in the model by adding X and Y coordinates as a smooth term (Dormann et al., 2007). The model was fitted with the function gamm4 from the R-package gamm4 (Wood, 2017).

3. Results

Mangrove extents changed considerably between 1984 and 2021 (from 738 km^2 to 895 km^2), with significant gains in some areas, typically inland, and losses mostly along the coast (Fig. 1). The largest losses were associated with coastline erosion, as clearly illustrated on the island of Maracá (Fig. 1, middle right panel). Areas in the north of the study area were relatively stable compared to those in the south nearer the mouth of the Amazon River.

Over the 38-year period, there was a positive growth in the extent of mangrove area (Fig. 2), with a gain of 1105 km^2 and a loss of 963 km^2. The overall net gain was 157 km^2 (Fig. 2).

When excluding the areas of coastline erosion and accretion, mangrove changes were more relevant at the inland edge, typically with an expansion following the edges of developing rivers and channels (Fig. 3).

Mangrove cover showed a significant increase over time, across all distance classes (Fig. 4; Table 1). There is a noticeable acceleration of mangrove growth between 1995 and 2000. The largest increase of mangrove cover was observed in the areas further than 9 km from the shoreline (Fig. 4).

Gauge data recorded near the northern edge of the study area (Cayenne) show an acceleration in sea level rise within the timeframe that mangrove accelerated expansion was observed (Fig. 5). This acceleration of sea level rise was also evident at Florida's gauge station of Vaca Key, which is the nearest station presenting a nearly complete dataset for the timeframe studied (Fig. 5).

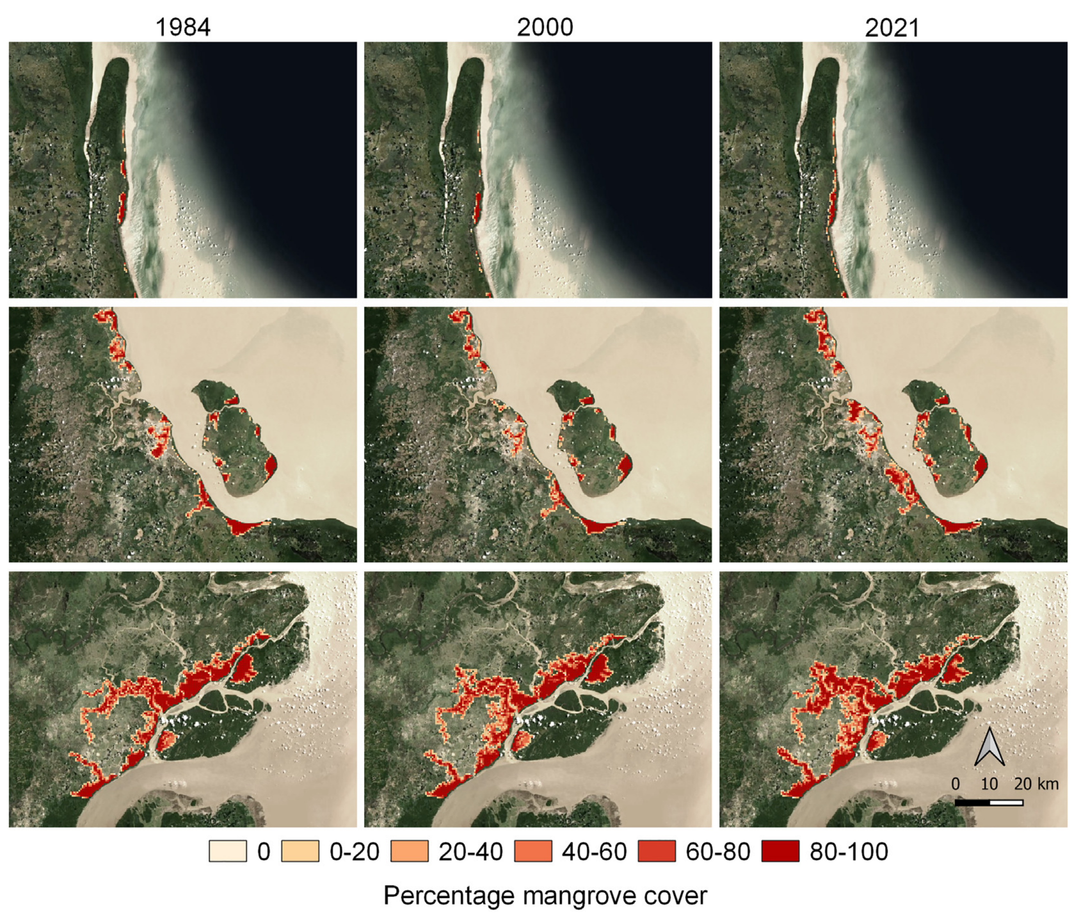

Fig. 3. Percentage mangrove cover per 500 × 500 m grid cell in 1984, 2000 and 2021 in three sections of the study area (locations are shown in squares in Fig. 1). Areas of coastline erosion and accretion were excluded. Background image – ESRI Satellite (ArcGIS/World_Imagery).

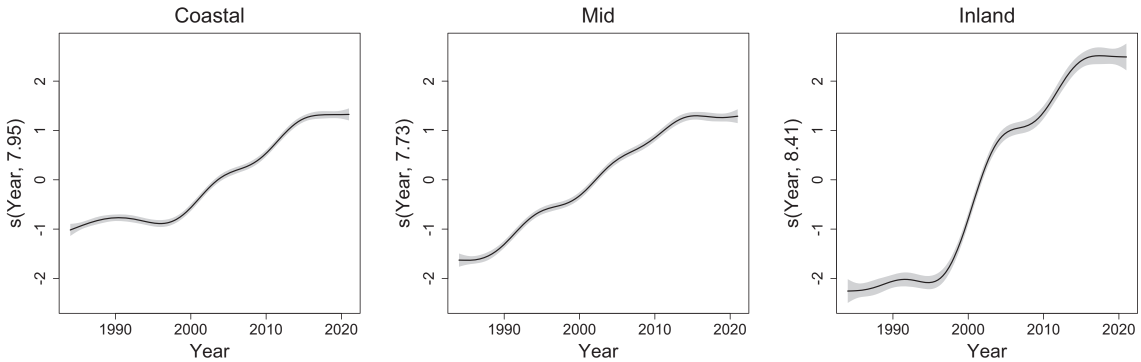

Fig. 4. Generalised Additive Mixed Model (GAMM) partial effects of year on the probability of mangrove cover in three distance classes to coastline: (1) Coastal (up to 3 km to the coastline), (2) Mid (3-9 km to the coastline), and (3) Inland (further than 9 km to the coastline). The model response variable was binomial, assigned to 1 if mangrove is present and 0 if absent. Cell ID was included as a random factor in the model. Spatial autocorrelation was accounted for in the model by adding X and Y coordinates as a smooth term. Shaded areas represent 95% confidence intervals.

The visualization of Landsat imagery in the Lake Bom Nome area (approximately 10 km^2) confirmed anecdotal reports by local communities suggesting that this lake had become saline over the last decade. There was a clear expansion of tidal channel networks in this area, which came into contact with the Bom Nome lake between 2004 and 2014 (Fig. 6). Once connected to the channel, the concentration of tidal sediments in the lake gradually increased (Fig. 6).

4. Discussion

We show that mangroves along the Brazilian Amazonian coast of Amapá have been expanding inland over the last 38 years, coinciding with sea level rise measurements and climate change signatures. Mangroves have expanded significantly at various distances from the shoreline, however, the inland areas (further than 9 km from the shoreline) demonstrated the largest increase (Fig. 3). Importantly, mangrove expansion accelerated between 1995 and 2000 at coastal and inland distance classes (Fig. 3). While variations in salinity were not measured in the field, the patterns of mangrove growth observed in this study are evidence of large-scale saltwater intrusion in this region. The mangrove area gained (1105 km^2) and lost (963 km^2) over the 38 year period monitored has been systematic (in the inland direction) rather than cyclic, suggesting a link to sea level rise and possibly to an increased frequency of tidal surges due to tropical storms, which has been predicted by climate change models (Gilman et al., 2008; Herbert et al., 2015).

Sea level rise measured by a nearby tidal gauge shows a clear acceleration (Fig. 5), which matches the acceleration observed in the mangrove growth at coastal and most inland areas in the study area (Fig. 4). It should be emphasised that although important data are missing in the time series gauge data of the nearest station (Cayenne), a more robust dataset recorded in Florida also shows evidence of sea level rise acceleration particularly in the last couple of decades (Fig. 5). This data is congruent with the global sea level rise acceleration observed from the late 20th century (Kirwan and Megonigal, 2013; Nicholls and Cazenave, 2010). The strongest acceleration of mangrove growth was observed at distances further than 9 km from the coastline, and this growth was mainly observed around tidal channels (Fig. 3). Sea level rise would indeed be expected to affect the areas around tidal channels more than those near the outer shoreline because tidal channels are likely to spread through low areas inland while the outer shoreline typically has higher elevation due to wave action (Anthony et al., 2008; Knighton et al., 1991). Nevertheless, mangrove growth has also accelerated in the nearest 3 km of the shoreline (Fig. 4). The shoreline is under direct influence of the tidal edge and consequently supratidal barriers, such as dunes and high terrain, may break due to rising sea levels, allowing mangrove forests to expand. In contrast, mangrove growth at intermediate distances from the shoreline (3 to 9 km) did not accelerate, possibly because these are vast areas covering extents beyond the tidal channels, where the influence of sea level rise might be more indirect.

Table 1
Summary statistics of a binomial Generalised Additive Mixed Model (GAMM) relating the probability of mangrove occurrence with time for three distance classes from coastline: (1) Coastal (up to 3 km to coastline), (2) Mid (3–9 km to coastline), and (3) Inland (farther than 9 km to coastline). Each grid cell was assigned 1 if mangroves were present and 0 if absent. Cell ID was included as a random factor. Spatial autocorrelation was accounted for in the model by adding X and Y coordinates as a smooth term. edf – estimated degrees of freedom.

Model smooth terms	edf	χ^2	*P*-value
s(Year): coastal	7.954	3779.2	<0.001
s(Year): mid	7.734	4168.7	<0.001
s(Year): inland	8.408	3568.3	<0.001
s(X, Y)	27.725	689.5	<0.001

Two local coastal freshwater lakes have increased in salinity in recent years, which also indicates large-scale saltwater intrusion in this region. High salinity values were measured in Lake Piratuba in April 2016 (da Cunha and Sternberg, 2018), and local villagers have reported a dramatic transformation of Lake Bom Nome over the last decade, with the lake currently being contaminated with marine sediments and under the influence of tidal regimes (observed by CDS). Freshwater fish species have been replaced by marine species and the original vegetation around the lake has been replaced by saline tolerant plant species (reported by villagers). These reports match the evidence acquired by satellite imagery, showing a systematic growth of tidal channels that eventually connected the lake to the ocean between 2004 and 2014 (Fig. 6).

It should be noted that, in addition to sea level rise, other factors might have contributed to saltwater intrusion in this region. Water buffalo farming is the most important economic activity in this region and buffalo grazing and paths can change the hydrology of wetlands significantly (Bowman et al., 2010). Water buffalo farming has been suggested as the main cause for the opening of two large channels at the southern end of our study area (Silva dos Santos et al., 2018), contributing to increased saltwater intrusion via the channels. Further to this, small tidal channels are sometimes opened to facilitate travelling with canoes and small boats between distant human settlements.

Villagers around the Bom Nome lake have reported a severe depletion of natural resources that they depend on. The new marine fish species are less abundant and less valuable compared to the freshwater species found previously. Safe drinking water is difficult to access, and local populations are enduring difficult living conditions. Water buffalo farmers have also reported the presence of saltwater in pastures and consequently have had to travel larger distances with their herds. Similar social consequences have been reported around the world in areas affected by saltwater intrusion (Tully et al., 2019).

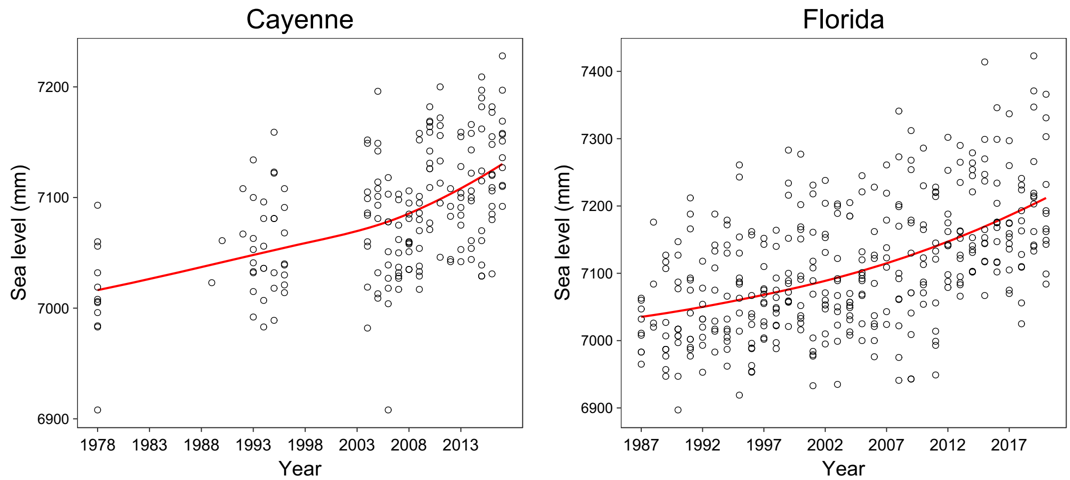

Fig. 5. Sea level rise recorded at tide gauges in Cayenne (Ilet la Mer, 4.8938 N, 52.1905 W), close to the Northern end of the study area and in Florida (Vaca Key, 24.7117 N, 81.1050 W), which presents a more complete time series. Datapoints represent monthly means of gauge sea level measurements, retrieved from the Permanent Service for Mean Sea Level database (https://www.psmsl.org). Loess smoothed trendlines are represented in red.

Importantly, the conversion of freshwater into saltwater environments and the saltwater intrusion in the Brazilian Amazon region is a potential threat to one of the largest carbon storage regions in the world (Gumbricht et al., 2017). Increased salinity and inundation of freshwater ecosystems cause vegetation die-off of salt-sensitive species, leading to the release of carbon (Charles et al., 2019). Further carbon is lost through the erosion of peat occurring from sea level rise and storm surges (Charles et al., 2019; Wong et al., 2013). However, these impacts may be mitigated in the long-term by the expansion of mangroves, which are highly effective at sequestering carbon (Charles et al., 2019). This study highlights that large-scale changes are occurring in remote areas of the planet with important carbon storage value, and the carbon balance of these changes are yet unknown.

The mangrove growth trend in the study area is likely to continue and accelerate in the future. Climate change models predict that sea level rise will continue to accelerate (Chang et al., 2011), and therefore saltwater intrusion is likely to become more prevalent in low-lying coastal regions, such as Amapá, and around the world (Wong et al., 2013). Saltwater intrusion and associated ecosystem changes has also been identified in other parts of the world – Florida, USA (Wan et al., 2015), Po Valley, Italy (Barbarella et al., 2015), Bangladesh (Mahmuduzzaman et al., 2014) and Northern Territory, Australia (Mulrennan and Woodroffe, 1998). As the speed of sea level rise and saltwater intrusion accelerates, freshwater ecosystems and coastal populations will have to adapt quickly if they want to withstand these rapid changes (Grieger et al., 2019).

Strategies to mitigate and adapt to the impacts of saltwater intrusion on coastal ecosystems and societies should encompass objectives of environmental, social, and economic sustainability (Gilman et al., 2006). An alternative sustainable approach to the construction of shoreline defences, such

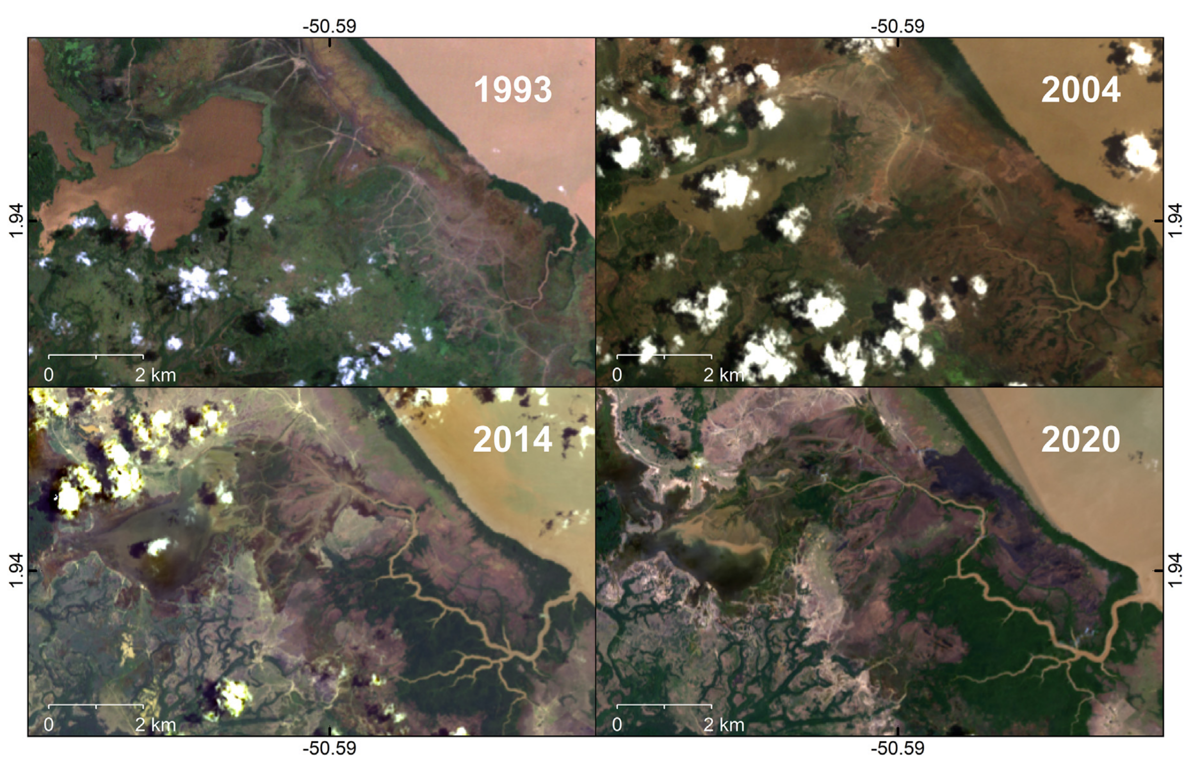

Fig. 6. The salinisation process of Lake Bom Nome (1.949° N, −50.624° E). Images are true colour composites of Landsat 5 (1993 and 2004) and Landsat 8 (2014 and 2020).

as sea walls, levees, and dikes, which generally have negative impacts on nearby coastlines (Bi et al., 2014), is enhancing natural systems at an ecosystem scale (Colls et al., 2009). Protected area networks may be used to maintain the resilience of healthy coastal wetlands (Leo et al., 2019). In addition, the management of coastal wetlands needs to include local and indigenous traditional knowledge and engage community stakeholders in planning, decision-making, and monitoring (Gilman et al., 2006).

5. Conclusions

This study has identified large-scale saltwater intrusion along the coastline of Amapá, indicated by accelerated inland migration of mangroves. The saltwater intrusion in this region is likely driven by sea level rise and possibly increased frequency of storm surges but other localised anthropogenic impacts such as buffalo grazing may also play an important role. An understanding of historical trends is needed to predict future impacts and inform adaptive management strategies. Coastal wetlands are undergoing rapid change and management strategies need to keep pace with this rate of change in order to effectively maintain healthy wetlands. Ongoing monitoring, using tools such as satellite imagery, as shown in this study, is key to ensuring that effective management strategies are established.

CRediT authorship contribution statement

Lola Laurie Baker Visschers: Conceptualization, Methodology, Formal analysis, Writing –original draft. Carlos D. Santos: Conceptualization, Formal analysis, Writing – review & editing. Aldina M. A. Franco: Conceptualization, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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