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Mangroves of the Tropical Southwestern Atlantic

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Abstract

The "Mangroves of the Tropical Southwestern Atlantic" (TSA) is a regional ecosystem subgroup (level 4 unit of the IUCN Global Ecosystem Typology). It includes the marine ecoregions of Eastern Brazil, Northeastern Brazil, including the Fernando de Noronha Archipelago and Rocas Atoll. The TSA mangroves had a mapped extent of 1719.7 km^2 in 2020, representing 1.2% of the global mangrove area. The flora is characterized by four true mangrove species: *Rhizophora mangle*, *Avicennia schaueriana, A. germinans* and *Laguncularia racemosa*. They provide essential biological functions such as supplying food resources to nearby settlements, including fish, crabs, mussels, and prawns, as well as timber and coastal defence. Furthermore, these environments promote ecological interactions with adjacent coral reefs. The climate ranges from humid to semi-arid along the Brazilian coast, which influences the diversity of geological environments and forest structures in the mangrove ecosystem. Despite their ecological and socio-economic importance, mangroves are at risk from conversion to aquaculture, salt panning, coastal infrastructure development, and oil pollution. Mangroves have also been significantly impacted by two recent major disasters: the 2015 collapse of the Mariana dam, which released a vast quantity of iron mining waste, and an extensive oil spill along the Brazilian northeast coast in 2019.

As of today, the TSA mangrove ecosystem area has undergone a net loss of -3.7% since 1996. If this trend continues an overall change of -6.7% is projected over the next 50 years. Furthermore, under a high sea level rise scenario (IPCC RCP8.5) ≈10.5% of the TSA mangrove area would be submerged by 2060. Moreover, 2.2% of the province's mangrove ecosystem is undergoing degradation, with the potential to increase to 6.6% within a 50-year period, based on a vegetation index decay analysis. Overall, the Tropical Southwestern Atlantic mangrove ecosystem is assessed as **Least Concern (LC).**

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Mangroves; Red List of ecosystems; ecosystem collapse; threats.

Ecosystem classification:
MFT1.2 Intertidal forests and shrublands

Assessment's distribution:
Tropical Southwestern Atlantic Province

Summary of the assessment:

Criterion	A	B	C	D	E	Overall
Subcriterion 1	**DD**	**LC**	**DD**	**DD**		
Subcriterion 2	**LC**	**LC**	**LC**	**LC**	**NE**	**LC**
Subcriterion 3	**DD**	**LC**	**DD**	**DD**		

LC: Least Concern, DD: Data Deficient, NE: Not Evaluated

Mangroves of the Tropical Southwestern Atlantic

LC

1. Ecosystem Classification

IUCN Global Ecosystem Typology (version 2.1, Keith *et al.* 2022):

Transitional Marine-Freshwater-Terrestrial realm

MFT1 Brackish tidal biome

MFT1.2 Intertidal forests and shrublands

MFT1.2_4_MP_14 Mangroves of the Tropical Southwestern Atlantic

IUCN Habitats Classification Scheme (version 3.1, IUCN 2012):

1 Forest

1.7 Forest – Subtropical/tropical mangrove vegetation above high tide level *below water level*[1]

12 Marine Intertidal

12.7 Mangrove Submerged Roots

Mangrove forest in Bahia dominated by the red mangrove, Rhizophora mangle (Photo credit: Arimatéa C. Ximenes).

[1] Note on the original classification scheme. This habitat should include mangrove vegetation below water level. Mangroves have spread into warm temperate regions to a limited extent and may occasionally occur in supratidal areas. However, the vast majority of the world's mangroves are found in tropical/subtropical intertidal areas.

2. Ecosystem Description

Spatial distribution

The mangroves of the Tropical Southwestern Atlantic (TSA) province include intertidal forests and shrublands of the marine ecoregions that extend across Brazil, including the eastern, northeastern, and southeastern coasts, the Fernando de Noronha Archipelago, and the Rocas Atoll (Figure 1). This province spans eight Brazilian states, and all state capitals, except for Teresina in Piauí State, are in coastal areas associated with mangrove ecosystems. The TSA province stretches from the Igaraçu River (2°51' S), a distributary of the Parnaíba River, located in the northern part of Piauí within the Northeast region of Brazil, to Cabo Frio (22°57' S), in Rio de Janeiro State, in the Southeast region of Brazil. The TSA boundary is situated adjacent to the Parnaiba Delta, extending along the northeastern coast to the São Francisco River. Within this region, the coastal area is characterised by smaller watersheds, which often exhibit intermittent flows towards the ocean (Pinheiro *et al*., 2016; Short and Klein, 2016). The Jequitinhonha, Caravelas, and Doce rivers span from the São Francisco River to the southeastern boundary of this province, namely along the Atlantic East Coast (Dominguez, 2006). The Paraiba do Sul River is located at province's southern boundary. This river serves as a conduit for the extensive watershed located between the states of São Paulo, Minas Gerais, and Rio de Janeiro. The area is characterised by a significant level of governance conflicts concerning water resources. The Paraíba do Sul River mouth is in the district of Atafona (Latitude 30°), which is in the municipality of São João da Barra in the Northern Fluminense Region of Rio de Janeiro. This river has a significant influence on the ecology of mangrove forests as a source of freshwater, sediments, and nutrients, and a means of propagule dispersion.

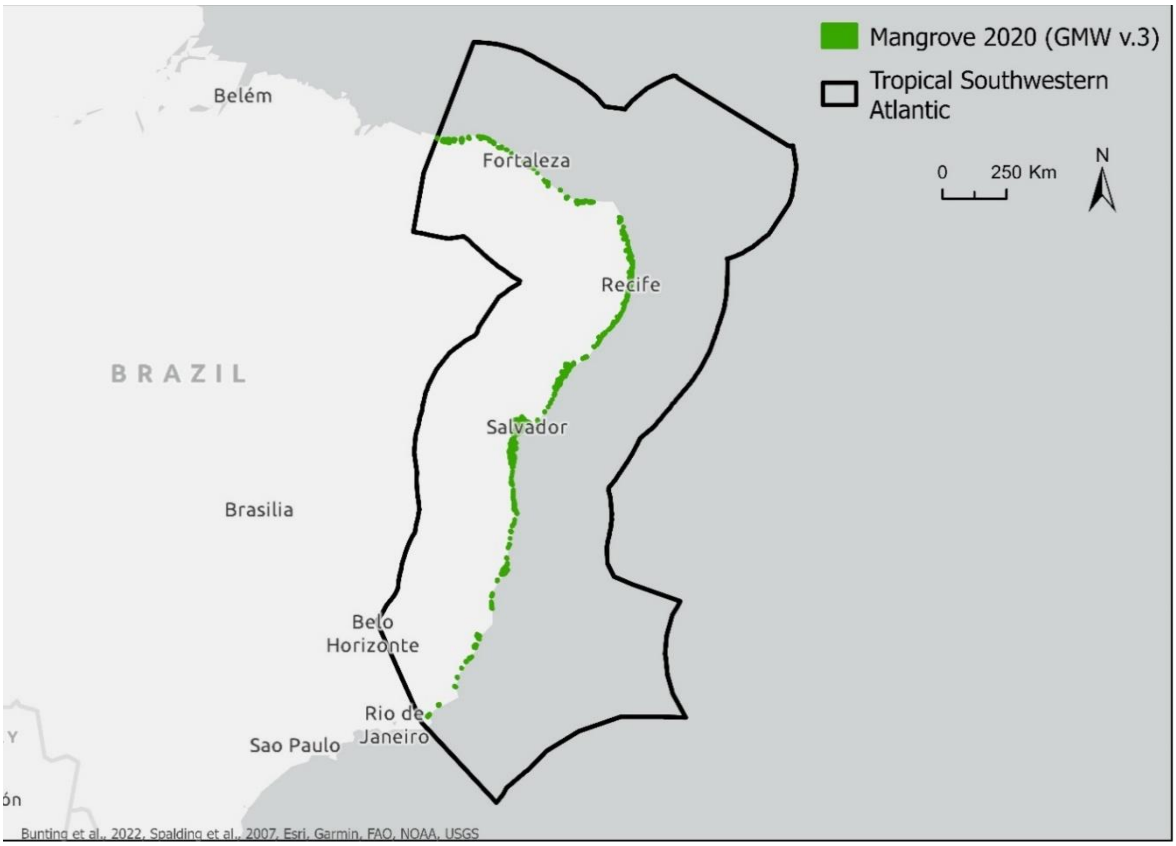

Figure 1. The mangroves of Tropical Southwestern Atlantic (TSA) province.

The entire area of mangroves along the Brazilian coast includes not only the TSA but also the North Brazil Shelf, and Warm Temperate Southwestern Atlantic ecoregions. There are different estimations of the total area of mangroves in Brazil. ICMBio reported an estimated area of around 13,989 km² (ICMBio, 2018). In contrast, the MapBiomas Collection v.8.0 estimated the Brazilian mangrove area at 10,344 km² in 1996, increasing to 10,370 km² in 2020 -a net increase of 26 km², or 0.25% over the same period (Diniz *et al.*, 2019). However, it is crucial to exercise caution when drawing conclusions about the reported regeneration of Brazilian mangroves over the past three decades (Diniz *et al*., 2019). The Global Mangrove Watch (GMW) v3.0 dataset (Bunting *et al.*, 2022), estimated the Brazilian mangrove area at 11,474 km² in 1996, decreasing slightly to 11,414 km² in 2020 – a net loss of approximately 60 km², or -0.5%. Regarding the TSA specifically, the GMW v3.0 reported that the mangrove area was 1,786.44 km² in 1996, decreasing to 1,719.7 km² in 2020—a net change of -66.74 km², or -3.7%, over the same period.

Biotic components of the ecosystem (characteristic native biota)

The mangroves of the Tropical Southwestern Atlantic (TSA) province include four of the most common true mangrove species in the Atlantic East Pacific (AEP) biogeographic region: *Avicennia schaueriana* Stapf and Leechm. ex Moldenke, *Avicennia germinans* (L.) L., *Rhizophora mangle* L., and *Laguncularia racemosa* (L.) C.F.Gaertn. (IUCN, 2022).

Furthermore, within the TSA province, coastal and inland habitats are characterised by the presence of common associated mangrove species, notably *Conocarpus erectus.* L. Tomlison (2016) classified *C. erectus* as a mangrove associate due to the absence of the morphological and biological traits typically associated with true mangroves, including pneumatophores and vivipary. Additionally, this species inhabits communities that are farther inland.

The southern limit of the TSA province marks the latitudinal boundary of *Avicennia germinans* in eastern South America (Schaeffer-Novelli *et al.*, 1990; Lacerda *et al.*, 2022a) and the northern boundary of the TSA coincides with the distribution limits of *R. racemosa* and *R. harrisonii* (Schaeffer-Novelli *et al.* 1990); therefore, these species do not occur within the TSA province. Brazil's mangroves are mainly distributed in humid regions; however, there are a few outliers in dry sub-humid areas along the border of Piauí and Ceará, as well as in semi-arid areas of Rio Grande do Norte, which extend just beyond the geographical limits of *R. racemosa* G. Mey. and *R. harrisonii* Leechm. (Ximenes *et al.*, 2016). According to Ximenes *et al.* (2016), the geographical distribution of these two *Rhizophora* species may be attributed to the occurrence of the driest seasons.

Laguncularia racemosa is the only mangrove species found within the insular mangroves of the Fernando de Noronha archipelago on the Atlantic Ocean coast (Guimarães *et al*., 2024). This species has been established in the region for approximately 8,000 years, since the Middle Holocene era during which this species thrived (Guimarães *et al*., 2024).

Laguncularia racemosa thriving on coastal rocks in Bahia, Brazil; this species demonstrates remarkable adaptability to harsh coastal environments (Photo credit: Arimatéa C. Ximenes).

In the transition zones between mangroves and other coastal ecosystems, such as hypersaline tidal flats (apicum ecosystems), dunes, and restingas, various plant species occur, including *Batis maritima*, *Portulaca oleracea*, *Salicornia sp*., and *Spartina sp*. (Camargo Maia and Coutinho, 2012; Costa *et al.*, 2014; Reis-Neto *et al.*, 2019). The Brazilian Northeast region frequently exhibits tidal estuaries with hypersaline conditions, where salinity gradually increases from the flood fringe to the upland (Costa *et al*., 2014; Lucena, 2012).

The dynamic transition zone between mangroves and apicum in Rio Grande do Norte State, which is a feature in this province (Photo credit: Arimatéa C. Ximenes).

In Brazilian mangroves, *Avicennia sp.* mostly flourish in habitats with high salinity, in contrast to *Rhizophora mangle* that typically occupies regions with lower levels of salinity (Soares *et al*., 2008; Ximenes *et al*., 2016; Pascoalini *et al*., 2019; Pellegrini *et al*., 2020; Tognella *et al*., 2020).

Fringe mangrove forest in Itacaré, Bahia dominated by red mangrove (Rhizophora mangle) (Photo credit: Arimatéa C. Ximenes).

Although mangrove habitats may have lower tree species diversity compared to other forest ecosystems globally, they are ecologically significant and provide essential support for a rich variety of organisms, including a wide range of bacteria, lichens, and fauna, many of which are still largely unknown to science. Historically, research in Brazil has concentrated on the northern and southeastern coastal areas, close to major urban centres, academic institutions, and established mangrove research groups. However, recent governmental policies have facilitated the expansion of research and educational institutions into previously neglected areas, though certain groups, such as the Teredinidae family and other mangrove infauna, remain unexplored (Silva and Tognella, 2013; Zamprogno *et al*., 2016). The Caatinga and Atlantic Tropical Forest biomes, known for their high biodiversity and endemism, are often connected to the mangrove forests in the TSA province (Rezende *et al*., 2018). Consequently, we expect a significant increase in the identification of bird and mammal species that reside within these mangrove ecosystems.

Among those already studied, the Largetooth sawfish (*Pristis pristis*), is Critically Endangered (CR) with overfishing and exploitation of spawning aggregations resulting in rapid decline of their populations (Sadovy and Eklund, 1999; CITES, 2007). Overfishing also threatens the Atlantic Goliath Grouper (*Epinephelus itajara*) and coupled with deterioration of its natural habitat (Bertoncini *et al*., 2018) has resulted in a Vulnerable (VU) classification on the IUCN Red List. As a keystone species, its decline has the potential to produce produce disproportionate effects in the ecosystem, emphasizing the immediate need for conservation efforts. Illegal wildlife trade is also an increasing pressure on the survival of species such as the Longsnout seahorse (*Hippocampus reidi*). This species is found in mangroves in Pernambuco (Borges *et al*., 2023), Ceará (Valentim *et al*., 2023) and in the northern coast of Espírito Santo (Tognella, pers. obs.). Coupled with habitat degradation, the illegal trade of this species has resulted in an evaluation of Near Threatened (NT) by the IUCN Red List (Oliveira and Pollom, 2017) and is listed as "Vulnerable" in the Brazilian national red list of threatened species, as determined by the Ministry of Environment and Climate Change of Brazil (MMA, 2022).

Tanks holding live mangrove crabs (Ucides cordatus; Left image) and blue land crabs (Cardisoma guanhumi; Right image), ready to sell to customers (Photo credit: Arimatéa C. Ximenes).

The mangrove fauna within the TSA province represents a crucial economic asset for traditional communities. Notably, the commercial trade of species such as *Ucides cordatus* (mangrove crabs) and *Cardisoma guanhumi* (blue land crabs) provides crab-harvesters with a subsistence income (Firmo *et al*., 2012). For example, traditional communities in Conceição da Barra, Espírito Santo rely on the trade of *Ucides cordatus* for financial support. Sales of six crabs for USD 5 are usually arranged by women and this is a vital activity supporting the economies of small villages. Mangrove-associated molluscs constitute another significant harvested taxon; however, comprehensive economic data pertaining to their stocks and commercial exploitation remain limited.

The mangrove ecology of the TSA province is enriched by the presence of the Caatinga and Atlantic Forest terrestrial biomes, and the Abrolhos Bank coral reef. The two forests support a wide variety of mammals, birds, amphibians, reptiles, and freshwater species which frequently inhabit the nearby mangroves to foraging, find shelter, and for breeding. Similarly, the Abrolhos bank provides access to the mangroves to many key marine species such as lobsters, turtles, and parrot fish. Together this combination of ecosystems generates greater species diversity than is usually encountered in mangrove forests.

Mangroves along a seasonal river in Ilheus, Bahia, featuring Avicennia schaueriana as the dominant species. The forest physiognomy exhibits signs of hydric stress (Photo credit: Mônica M. P. Tognella).

Abiotic Components of the Ecosystem

Interactions between catchment land cover, landscape position, rainfall, hydrology, sea level, sediment dynamics such as erosion and accretion, storm-driven processes, and disturbance by pests and predators influence mangrove species distributions. Rainfall and sediment supply from rivers and tidal currents promote mangrove establishment and persistence, while waves and large tidal currents destabilise and erode mangrove substrata. High rainfall reduces salinity stress and increases nutrient loading from adjacent catchments, while tidal flushing also regulates salinity.

Based on historical climate data obtained from the WorldClim database v1, Ximenes *et al*. (2016) observed that Macau, located in Rio Grande do Norte, receives the least rainfall along Brazil's mangrove coastline, averaging only 600 mm annually. Similarly, Rio de Janeiro has low annual precipitation near Arraial do Cabo, with an average of 870 mm. Nevertheless, several locations in Ceará, Bahia and Espírito Santo also have very modest levels of annual precipitation, with measurements of 937 mm, 990 mm, and 1003 mm, respectively (Ximenes *et al.*, 2016). The low rate of precipitation and the high evapotranspiration associated with the geomorphology contribute to the formation of salt flat patches along the edges and within the mangrove forests (Schaeffer-Novelli *et al*., 1990). Changes in estuarine sediment dynamics and freshwater influx lead to modifications in mangrove forest structure as well as shifts in species zonation and distribution patterns. The regions in Brazil that exhibit the lowest levels of moisture in mangrove ecosystems are situated in the northeastern states of Piauí, Ceará, and Rio Grande do Norte (Ximenes *et al.*, 2016). In the driest quarter of the year, precipitation varies significantly across regions, ranging from 1 mm in Camocim, Ceará, to a peak of 466 mm in Bahia, specifically in the Camamu/Maraú area. This demonstrates a notable increase in precipitation as one moves southward, highlighting the regional climatic variations within Brazil (Ximenes *et al*., 2016). Camamu/Maraú (Bahia), Suape (Pernambuco), and Costa das Algas (Espírito Santo) have unique coastal mangroves associated with reef flats in Bahia and Pernambuco, and with cliffs and rock platforms characterised by huge movable dune fields, as documented by Camargo *et al.* (2006).

Mangroves in Costa das Algas (ES) where Avicennia schaueriana grows on reef fragments, providing protective support for Langularia racemosa and Rhizophora mangle located inland (Photo credit: Mônica M. P. Tognella).

Short and Klein (2016) proposed a geomorphological classification that divides the TSA province into three distinct regions. The first region is the Northern tide-modified barrier coast, spanning from São Luís in Maranhão state to Touros in Rio Grande do Norte state. The second region is the Northwestern wave-dominated beachrock coast, extending from Touros to Coruripe in Alagoas state. The third region is the Eastern wave-dominated deltaic coast, encompassing the area from the São Francisco River in Alagoas state to Cabo Frio in Rio de Janeiro state.

The influence of the North Brazilian Current and easterly trade winds on the tide-modified Barrier Coast has been described by Stramma *et al.*, (1995). These oceanic forces drive longshore sediment movements, resulting in the formation of transgression dune fields, as discussed by Short and Klein (2016) and Ward and Lacerda (2021). The second zone is characterised by the impact of the Brazilian Current and the southeast trade winds. In the wave-dominated beachrock sector, the rivers exhibit a low volume of water flow, and hence, sediment inputs to the ocean are minimal (Short and Klein, 2016). In some areas of the TSA region, the presence of bedrock and beachrock has served as a protective mechanism for mangrove forests against the forces of tides and wave energy, thereby facilitating the development of coastal mangrove ecosystems. The third zone within the TSA is characterised by deltas and tiny lagoons, which are home to rivers that exhibit the largest discharge into the Atlantic Ocean along the Brazilian coast (MMA, 2006; Short and Klein, 2016). In the Tropical Southwestern Atlantic, the tidal amplitude varies from macro-tidal (6 m) to micro-tidal (1.5 m) as one moves from the northern to the southern regions (Ward *et al.*, 2016).

Key processes and interactions

The unique characteristics of mangroves and their interconnectedness with other ecosystems in this province give rise to distinct processes and interactions, such as nutrient cycling, habitat provision for diverse species, and the maintenance of coastal stability, all of which contribute to the overall biodiversity of the region. Mangroves are structural engineers. They have features like pneumatophores, salt excretion glands, viviparity, and propagule buoyancy that help them survive and reproduce in substrata that are saline, mobile, and experience tidal flooding (Tomlinson, 2016). Mangroves produce large amounts of detritus (e.g., leaves, twigs, and bark), which is either buried in waterlogged sediments or consumed by crabs and gastropods. This detritus is further decomposed by meiofauna, fungi, and bacteria.

Sediment deposition, freshwater flow, and salinity variations in aquatic environments impact the physical characteristics of mangrove forest ecosystems. These factors contribute to variability in plant species diversity and relative abundance within estuarine areas, ultimately shaping the composition and structure of the forest community (Tognella *et al.,* 2020, 2021). For example, this variability can lead to habitats ranging from sparse, highly saline aggregations of a few plant species to dense thickets or biodiverse mangrove forests.

Mangroves influenced by tidal dynamics exhibit elevated flow velocities that surpass those of contaminants, resulting in a brief residence time for contaminants, eggs, and larval stages within the estuary. This rapid movement minimizes the accumulation of these materials, thereby limiting their potential impacts on local ecosystems. However, the capacity for marine pollutants to disperse over significant distances within the estuary and into adjacent coastal waters remains a concern (Lacerda *et al*., 2021; Tognella *et al*., 2021). Thus, the role of mangroves is crucial in regulating the exchange of materials at the land-ocean interface, particularly in mitigating the effects of pollutants. Local hydrology serves as a key driver of these exchanges, influenced by hydrological connectivity and the localized effects of global climate change (Lacerda *et al*., 2022a; Lacerda *et al*., 2022b). Understanding these relationships is essential for assessing the ecological functions of mangroves and their contributions to coastal ecosystem health.

Mangrove ecosystems also serve as major blue carbon sinks, incorporating organic matter into sediments and living biomass (Taillardat *et al*. 2018). Nóbrega *et al.,* (2019) investigated the impact of aquaculture on carbon storage in mangrove ecosystems. Their findings indicated that aquaculture activities can lead to changes in carbon storage patterns, which can have implications for the overall carbon balance of these ecosystems. Shrimp farm effluents could potentially have a negative impact on mangroves' ability to sequester carbon in soil. This is mostly because the nutrient-rich wastewater from aquaculture feeds a population of bacteria that breaks down organic matter more quickly (Lacerda *et al.,* 2021). A reduction in legislation protection in Brazil, in addition to the impacts on carbon sequestration from aquaculture activities and related contaminant release, could potentially result in the loss of up to 600,000 ha of mangroves, leading to a decrease in sequestration rates of up to 11.175 tC ha yr^{-1} (Ward and Lacerda, 2021). Ward *et al*. (unpublished in review) have reported soil carbon sequestration rates in fringing mangroves of 134-3248 g m^2 yr^{-1} for mangroves in the semi-arid northeast of Brazil (Ceara), and Hatje *et al*. (2021) reported slightly lower levels for mangroves in Bahia (649-904 g m^2 yr^{-1}), with those highest levels in Ceara being linked to extreme urban source organic matter inputs.

Research conducted in the southeastern region of Brazil (RJ) has revealed that urban mangroves exhibit carbon sequestration rates between 4 and 8 times higher than non-urban mangroves (Sanders *et al.*, 2014). Additionally, Rovai *et al*. (2022) emphasised that Brazilian mangroves possess significant elevated carbon sequestration rates compared to global averages that can help in mitigate climate change.

3. Ecosystem Threats and vulnerabilities

Main threatening process and pathways to degradation

There are many factors that result in mangrove degradation and loss, including aquaculture, urbanisation, coastal development, excessive harvesting, and pollution from household, industrial, and agricultural activities (Goldberg *et al*. 2020).

Furthermore, the presence of mangrove forests in intertidal zones makes them susceptible to anticipated sea-level rise due to climate change (Ward and Lacerda, 2020). This will result in the loss of mangrove area and would force a shift in the geographical distribution of these forests inland. However, inland migration is constrained by urbanisation and other human activities in the surrounding mangrove areas and such as the damming of rivers, which limits the ability of this ecosystem to adapt to future change (Ward *et al.*, 2023).

Two significant events in the TSA province have further exacerbated environmental concerns: the collapse of the Fundão dam in 2015 (Tognella *et al*., 2020) and an oil spill along the northeast coast that started in late August 2019 (Magris and Giarrizzo, 2020). These events have wide ranging biotic implications, for example, many crab populations are now declining, with females exhibiting changes in fertility timing and depositing fewer eggs (Lima *et al.*, 2023; Porto *et al.*, 2021).

Pollution

Brazil has a lengthy history of oil disasters impacting mangrove habitats (Lassalle *et al*. 2023). In 2019, an unprecedented oil spill affected the coast of Brazil, making it one of the biggest environmental catastrophes in the world and causing a series of interconnected socioeconomic and ecological problems (Magris and Giarrizzo 2020). In late August, a layer of crude oil spread across almost 3,000 kilometres of shoreline, impacting 11 Brazilian states from Maranhão to the northern coastal region of Rio de Janeiro (IBAMA, 2020). Despite the combined efforts of government agencies, local communities, and environmental organisations, the identification of the exact cause and responsible parties for the spill remains elusive, making spill containment and cleanup challenging. This tragic event highlighted the vulnerability of marine ecosystems and the urgent need for enhanced safeguards and cooperative efforts to protect coastal areas.

Ports and densely populated areas face a persistent pollution problem due to the release of municipal sewage and oil spills produced by naval shipping in their immediate vicinity. One such example is the use of antifouling paint. Previous widespread use and subsequent leaching of tin, led to the incidence of imposex in molluscs (Costa *et al.*, 2014). Despite a global ban, Brazil continued to manufacture antifouling paint, which was applied on small wooden boats navigating through mangrove areas. The utilisation of this method has been demonstrated to lead to the development of intersex traits in *Littoraria angulifera*, as evidenced by Costa *et al*. (2014).

Laguncularia racemosa (white mangrove) in Pernambuco State, Brazil, coated with crude oil from the 2019 oil spill (Photo credit: Clemente Coelho-Jr.).

Pollution in a river bordered by mangroves in Ceará State (Photo credit: Armando Reis-Neto).

Aquaculture

Deforestation is a major consequence of aquaculture development in mangrove ecosystems (Friess *et al.*, 2019). In Brazil, approximately 13% of mangrove forests have been lost to aquaculture (Hamilton, 2013). This figure is comparatively lower than in Southeast Asia (29.9%) and countries such as Vietnam (53%), Indonesia (48%), and Ecuador (40%) (Veettil *et al.*, 2019). However, the measurement of total area or the extent of area loss alone does not serve as a reliable indication of the overall health of an ecosystem. In northeastern Brazil, significant indirect impacts from shrimp pond effluents and the construction of water channels for supply and drainage have been linked to altered hydrological dynamics and increased nutrient and heavy metal inputs, among other impacts (For a review see Lacerda *et al.,* 2021).

A study comparing the Normalised Difference Vegetation Index (NVDI) metrics showed a 17% decrease in NDVI in the Jaguaribe River estuary, associated with a ~470% increase in aquaculture pond area. This expansion has significant implications for the overall functioning of mangrove ecosystems (Marins *et al.,* 2020). The vegetation index is influenced by several plant characteristics, such as pigment composition (Silva *et al*., 2020), water content, and carbon content and these characteristics may be modified followed changes to hydrological dynamics that occur during the excavation and installation of prawn ponds (Alatorre *et al.,* 2016). The reported decrease in NDVI was due to tree mortality and thinning of canopy associated with altered hydrological dynamics, as well as nutrient rich effluents (Marins *et al.*, 2020). Nutrient emissions associated with aquaculture operations in the semi-arid coast of Brazil are comparatively greater than those of other rural area activities such as agriculture, animal husbandry, solid waste disposal, urban runoff, and sewage (Lacerda *et al.* 2006). The increase in nutrients can cause eutrophication, the proliferation of microorganisms, such as microalgae, with subsequent oxygen depletion, impacting the functioning of mangrove ecosystems (Nóbrega *et al.*, 2014).

In addition, heavy metal emissions, particularly of bioavailable chemical species, can be high in shrimp farms in northeastern Brazil, indirectly impacting the adjacent mangrove (Lacerda *et al.,* 2021). The accumulation of trace metals within cultivation ponds can result in their subsequent emission to the nearby mangroves. These contaminants are commonly present as impurities in aquaculture supplies, including aquafeeds, fertilisers, lime, and chloride (Lacerda *et al.,* 2009, 2011). Heavy metals often found in shrimp pond effluents include copper, zinc, and mercury, which possess the potential to induce toxicity in species even at very low ambient concentrations.

Eolic farm bordering, salt pans and aquaculture ponds impacting and disturbing mangrove forest in Macau, Rio Grande do Norte (Photo credits: Arimatéa C. Ximenes, 2024)

Residential and infrastructure development

The history of coastal settlement in Brazil primarily began in coastal areas along the Tropical Southwestern Atlantic (TSA) province (Netto and Reis-Neto, 2023). This trend has led to urban and industrial expansion that suppresses mangrove ecosystems, highlighting the need for a deeper understanding of the ecological impacts of such development (Hamilton and Snedaker, 1984; Silva *et al*., 2001). Almost all major Brazilian cities in this region are situated in coastal environments, which has driven widespread utilization of mangrove wood for construction, fuel, and charcoal production, thereby impacting the ecological integrity of these vital ecosystems (Netto and Reis-Neto, 2023). Furthermore, the development of aquaculture over centuries has contributed to ongoing land-use changes, further influencing mangrove environments (Scott, 2013). The interplay between urban development and mangrove resource exploitation underscores the urgent need for sustainable management practices to protect these critical ecosystems.

The TSA province harbours several prominent cities in the Brazilian Northeast, predominantly situated in estuarine environments. Particularly within the sub-province known as Semiarid Equatorial Coast (SAE), urbanisation has extended into regions neighbouring the historical saltworks constructed along the river peripheries. The estuarine regions in the vicinity have been used for salt production since the 16th century (Costa *et al*., 2012).

The expansion of harbours and industrial infrastructure continues to have a significant influence on mangrove forests (Lacerda *et al.*, 2002). The establishment of the SUAPE port and the subsequent implementation of an industrial structure in Recife, located in the State of Pernambuco, resulted in significant earthwork and dredging activities. These interventions had a profound impact on the local hydrology, leading to vast modifications and causing substantial damage to the mangrove region (Braga, 1989). The ports significant threats to local mangrove remnants, as well as posing a regional risk to adjacent regions situated along ship routes. For instance, the increased shipping traffic and associated pollution can lead to habitat degradation and loss of ecosystem services, posing significant regional risks to adjacent areas along shipping routes.

A bridge dividing the mangrove forest in two fragments in Ceará State (Photo credit: Arimatéa C. Ximenes).

Mangrove forests impacted by human habitation and discarded garbage, Espírito Santo State (Photo credit: Mônica M. P. Tognella).

Mangroves transitioning between hypersaline tidal flats (apicum) and mangrove ecosystems, adjacent to coconut plantations, shrimp farms, and wind farms in Aracaú, Ceará, Brazil (Photo credits: Arimatéa C. Ximenes).

Dams

Multiple dams have been constructed across various watersheds in Brazil, resulting in a decrease in the volume of freshwater and silt reaching the coastal areas. Nonetheless, tides and currents have a significant effect on river flow, while waves facilitate the transportation of silt from the coastal region to the river (Souza *et al*., 2014, Tognella *et al.*, 2021).

The Mariana Dam disaster occurred on November 5, 2015, when the Fundão Dam had a catastrophic collapse. The dam contained around 50 million cubic metres of iron mining waste, which resulted in the dispersion of metallic tailings throughout the northern shore of Espírito Santo (Escobar, 2015). This event has caused significant ecological consequences, particularly affecting coastal ecosystems such as estuaries and mangroves (Tognella *et al.*, 2021). This catastrophe is still causing alterations in carbon absorption, photosynthetic pigments, photosynthetic efficiency, and the buildup of trace metals in flora and sediments (D'adazzio *et al.*, 2023).

In conjunction with these stressors, a significant number of hydrological basins experience flow disruptions due to the presence of dams. The upstream presence of dams reduces freshwater and sediment inflow into the lower estuary, thereby augmenting the transportation of sediments by tidal forces from the estuary mouth to the upper estuary. Along the coast of Ceará, the transportation process is affected by the dominant eastern winds and the movement of sediment along the coastline. This leads to alterations in the physical features of the mangrove environment, including its burial, alteration of freshwater flow, changes in salinity, reduced sedimentation, disruption of natural tidal cycles, and erosion of shorelines amongst other effects. In Ceara, dam construction has resulted in the expansion of mangroves in the upper estuary as a result of decreases in water levels, although the consequent decrease in sediment inputs has had a negative impact on those mangroves further downstream. The latter have become sediment-starved, which is reducing their ability to adjust to increases in sea level (Ward *et al.*, 2023).

Environmental protection

The Brazilian mangrove environment is safeguarded by a total of 93 conservation units, including both integral protection and sustainable use areas. The majority of these protected areas were established at the beginning of the current century. Additionally, there are 27 extractive reserves, often referred to as RESEX, which serve to preserve not only the ecosystem itself but also the indigenous communities that rely on it (ICMBIO, 2018). It is important to highlight that most of these conservation units and extractive reserves fall under the jurisdiction of the federal government because this governance structure plays a critical role in determining conservation policies and resource management strategies. Federal oversight ensures a coordinated approach to environmental protection, which is essential for the effective preservation of these areas and the sustainable use of their resources (Tognella *et al*., 2019).

The TSA province has a total of 29 coastal conservation units, out of which 7 are designated as Reserves Extractive (RESEX). The establishment of RESEX has provided traditional communities with the authority and means to ensure the sustainable management and preservation of mangrove ecosystems, which in turn support the provision of many valuable commodities and services.

Definition of the collapsed state of the mangrove ecosystem

Mangrove ecosystem collapse occurs when the tree cover of true mangrove species declines to zero, indicating total habitat loss. Ecosystem collapse may manifest through the following mechanisms: a) removal of the mangrove forest cover to replace to other land use; b) restricted recruitment and survival of true mangroves due to adverse environmental conditions, for example, alterations in rainfall, river inputs, waves, and tidal currents that destabilise and erode substrata, hindering recruitment and growth; c) shifts in rainfall patterns and tidal flushing alter salinity stress and nutrient loadings, impacting overall survival; d) simultaneous environmental catastrophes such as oil spills and collapses of dams.

Most river headwaters originate in regions characterised by high levels of precipitation. However, these rivers go through a semi-arid environment as they make their way towards the coast. In coastal areas, the combination of climate, limited topographic variation, and few drainage basins, leads to a reduced input of sediment (Dominguez, 2009). In addition, decreased levels of precipitation, coupled with increased urbanisation and agricultural activities, have resulted in a decline in the volume of freshwater that reaches mangroves throughout its journey towards the ocean. The prospective population growth in coastal cities and other mangrove areas in the TSA province is concerning and could lead to the ecosystem's collapse.

Climate change has the potential to modify hydrological patterns, leading to an anticipated rise in occurrences of severe climatic events such as droughts and storms. These effects are particularly pronounced in regions characterised by dry and semi-arid climates (Jennerjahn *et al.*, 2017).

Tragically, the environmental impacts of anthropogenic activities are leading to a decline in crab populations (Bromenschenkel and Tognella, 2020). The first large-scale impact on crab populations occurred following a fungal contamination in 2005 and 2006 (Boerger *et al*., 2005; Borger *et al*., 2007). This contamination resulted in increased mortality rates among crabs in numerous estuaries along the northeastern coast of Brazil, which had significant socio-economic consequences for crab harvesters (Firmo *et al*., 2012).

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Threat Classification

IUCN Threat Classification (version 3.3, IUCN-CMP, 2022) relevant to mangroves of the Tropical Southwestern Atlantic province:

1. Residential & commercial development
- 1.1 Housing & urban areas
- 1.2 Commercial & industrial areas
- 1.3 Tourism & recreation areas

2. Agriculture & aquaculture
- 2.2 Wood & pulp plantations
  - 2.2.1 Small-holder plantations
  - 2.2.2 Agro-industry plantations
- 2.3 Livestock farming & ranching
  - 2.3.1 Nomadic grazing
  - 2.3.2 Small-holder grazing, ranching or farming
  - 2.4 Marine & freshwater aquaculture
  - 2.4.1 Subsistence/artisanal aquaculture
  - 2.4.2 Industrial aquaculture

3. Energy production & mining
- 3.1 Oil & gas drilling
- 3.2 Mining & quarrying
- 3.3 Renewable energy

4. Transportation & service corridors
- 4.1 Roads & railroads
- 4.2 Utility & service lines
- 4.3 Shipping lanes
- 4.4 Flight paths

5. Biological resource use
- 5.1 Hunting & collecting terrestrial animals
  - 5.1.1 Intentional use (species being assessed is the target)
  - 5.1.2 Unintentional effects (species being assessed is not the target)

- 5.2 Gathering terrestrial plants
    - 5.2.1 Intentional use (species being assessed is the target)
    - 5.2.2 Unintentional effects (species being assessed is not the target)
- 5.3 Logging & wood harvesting
    - 5.3.1 Intentional use: subsistence/small scale (species being assessed is the target [harvest]
    - 5.3.2 Intentional use: large scale (species being assessed is the target)[harvest]
    - 5.3.3 Unintentional effects: subsistence/small scale (species being assessed is not the target)[harvest]
- 5.4 Fishing & harvesting aquatic resources
    - 5.4.1 Intentional use: subsistence/small scale (species being assessed is the target)[harvest]
    - 5.4.2 Intentional use: large scale (species being assessed is the target)[harvest]
    - 5.4.3 Unintentional effects: subsistence/small scale (species being assessed is not the target)[harvest]
    - 5.4.4 Unintentional effects: large scale (species being assessed is not the target)[harvest]
    - 5.4.5 Persecution/control
    - 5.4.6 Motivation Unknown/Unrecorded

6. Human intrusions & disturbance

- 6.1 Recreational activities
- 6.2 War, civil unrest & military exercises
- 6.3 Work & other activities

7. Natural system modifications

- 7.1 Fire & fire suppression
    - 7.1.1 Increase in fire frequency/intensity
    - 7.1.2 Suppression in fire frequency/intensity
    - 7.1.3 Trend Unknown/Unrecorded
- 7.2 Dams & water management/use
    - 7.2.1 Abstraction of surface water (domestic use)
    - 7.2.2 Abstraction of surface water (commercial use)
    - 7.2.3 Abstraction of surface water (agricultural use)
    - 7.2.4 Abstraction of surface water (unknown use)
    - 7.2.5 Abstraction of ground water (domestic use)
    - 7.2.6 Abstraction of ground water (commercial use)
    - 7.2.7 Abstraction of ground water (agricultural use)
    - 7.2.8 Abstraction of ground water (unknown use)
    - 7.2.9 Small dams
    - 7.2.10 Large dams
    - 7.2.11 Dams (size unknown)
- 7.3 Other ecosystem modifications

8. Invasive & other problematic species, genes & diseases

- 8.1 Invasive non-native/alien species/diseases
    - 8.1.1 Unspecified species
    - 8.1.2 Named species
- 8.2 Problematic native species/diseases
    - 8.2.1 Unspecified species
    - 8.2.2 Named species
- 8.3 Introduced genetic material
- 8.4 Problematic species/diseases of unknown origin

    - 8.4.1 Unspecified species
    - 8.4.2 Named species
- 8.5 Viral/prion-induced diseases
    - 8.5.1 Unspecified "species" (disease)
    - 8.5.2 Named "species" (disease)
- 8.6 Diseases of unknown cause

9. Pollution

- 9.1 Domestic & urban waste water
    - 9.1.1 Sewage
    - 9.1.2 Run-off
    - 9.1.3 Type Unknown/Unrecorded
- 9.2 Industrial & military effluents
    - 9.2.1 Oil spills
    - 9.2.2 Seepage from mining
    - 9.2.3 Type Unknown/Unrecorded
- 9.3 Agricultural & forestry effluents
    - 9.3.1 Nutrient loads
    - 9.3.2 Soil erosion, sedimentation
    - 9.3.3 Herbicides & pesticides
    - 9.3.4 Type Unknown/Unrecorded
- 9.4 Garbage & solid waste
- 9.5 Air-borne pollutants
    - 9.5.4 Type Unknown/Unrecorded
- 9.6 Excess energy
    - 9.6.2 Thermal pollution
    - 9.6.3 Noise pollution
    - 9.6.4 Type Unknown/Unrecorded

10. Geological events

- 10.3 Avalanches/landslides

11. Climate change & severe weather

- 11.1 Habitat shifting & alteration
- 11.2 Droughts
- 11.3 Temperature extremes
- 11.4 Storms & flooding
- 11.5 Other impacts (sea-level rise)

4. Ecosystem Assessment

Criterion A: Reduction in Geographic Distribution

Subcriterion A1 measures the trend in ecosystem extent during the last 50-year time window. Unfortunately, there is currently no national dataset that provides information for the entire target area (TSA) back to 1970. National-level data is available, but only from 1985 onward, and some studies focus on the 90's to 2020 period. At the subnational level, only Espírito Santo has data dating back to 1970. This current lack of comprehensive data across the region makes it difficult to accurately extrapolate the trend to 1970, as national data is not close enough to that date, and subnational data is not available for the entire TSA.

Reliable published sources with mangrove area estimates at both national and subnational levels were compiled (see Appendix 4). There are substantial discrepancies in the mangrove extent estimates (Diniz *et al*., 2019;

Bunting *et al*., 2022; Ximenes *et al*., 2023) and in the net area change calculated from these estimates for the entire mangrove area in Brazil (appendix 4). For example, based on Diniz *et al.* (2020) the net area change between 1996 and 2020 was 0.4%, while using Bunting *et al.* (2022) data, the net change for the same period was -0.5%. As a result, the overall change in mangrove area may exhibit either a positive or negative trend, depending on the source.

At the subnational level, the specific characteristics of each location influence the extent of net area change within the TSA. In Espirito Santo, historical mapping from 1970 to 2015 by IEMA shows a 16.6% decrease in mangrove area (2,273 km²), with an annual change rate of -0.36% (IEMA, 2023). In contrast, three distinct estuaries in the Ceara state, situated in the northeastern region of Brazil, saw an alteration in the extent of mangrove coverage from 1992 to 2011, ranging from an increase of 28.3% to 47.8% of mangrove coverage in these estuaries (Godoy and Lacerda, 2015). This expansion may be attributed to increased sedimentation along the river margins, due to decreased water flow from river dam construction (Ward *et al*., 2023). However, it is also possible that the loss of fringing mangroves located at the mouths of these estuaries is a result of the actions of waves, sea level rise, currents, tides, and various weather phenomena. Thus, it is important to ascertain the precise changes in the mangrove area within this province.

The reduction in mangrove extent has the potential to decrease the existence of natural buffer zones, hence impacting the resilience of ecosystems in the TSA region. In Espírito Santo State, these natural buffer zones, classified as fluvio-marine plain have been decreasing in an order of -0.4 % per year since 1970 (IEMA, 2023). This risk is accentuated in semi-humid estuaries (north coast of State) and in the metropolitan area of Vitória.

While the data from Espirito Santo indicates a large decline in the mangrove area, there is a lack of data for the entire TSA for the past 50 years. Therefore, the Tropical Southwestern Atlantic is considered as **Data Deficient (DD)** for subcriterion A1.

Subcriterion A2 measures the change in ecosystem extent in any 50-year period, including from the present to the future: The Tropical Southwestern Atlantic province mangroves show a net area change of -3.7% (1996-2020) based on the Global Mangrove Watch time series (Bunting *et al.*, 2022). This value reflects the offset between areas gained (+ 0.1 %/year) and lost (- 0.36 %/year), similar behaviour was observed in the mapping produced by the Institute of Environment and Water Resources from the state of Espírito Santo (IEMA, 2023). Applying a linear regression to the area estimations between 1996 and 2020 we obtained a rate of change of -0.2%/year (figure 2). Assuming this trend continues in the future, it is predicted that the extent of mangroves in the Tropical Southwestern Atlantic province will change by -7.0% from 1996 to 2046; by -10.2% from 1996 to 2070; but by -6.7% from 2020 to 2070. Given that these predicted changes in mangrove extent are below the 30% risk threshold, the Tropical Southwestern Atlantic mangrove ecosystem is assessed as **Least Concern (LC)** under subcriterion A2.

Mangroves of the Tropical Southwestern Atlantic	Area 2020* (Km^2)	Area 1996* (Km^2)	Net area Change (Km^2)	% Net Area Change	Rate of change (%/year)
	1719.7	1786.44	66.74	-3.7	-0.2%

** Details on the methods and references used to estimate the mangrove area in 1970 are listed in appendix 4. Total mangrove area in 2020 is based on the Global Mangrove Watch Version 3 (GMW v3.0) dataset.*

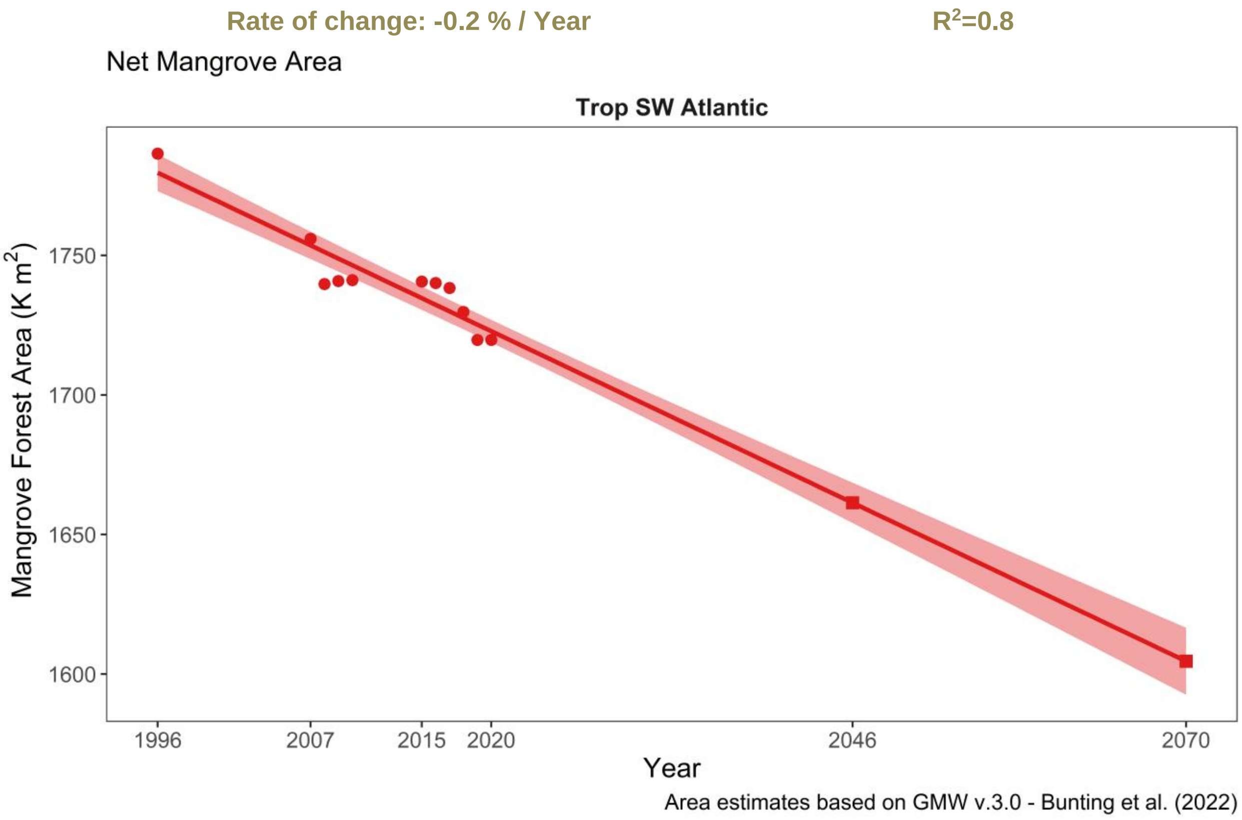

Figure 2. Projected extent of the Tropical Southwestern Atlantic mangrove ecosystem to 2070. Circles represent the province mangrove area between 1996 and 2020 based on the GMW v3.0 dataset and equations in Bunting *et al.*, (2022). The solid line and shaded area are the linear regression and 95% confidence intervals. Squares show the Tropical Southwestern Atlantic province predicted mangrove area for 2046 and 2070. It is important to note that an exponential model (proportional rate of decline) did not give a better fit to the data (R^2 = 0.8).

The possible future decline of mangrove areas in Brazil may also be substantially influenced by changing government policies. Historically, mangroves were subject to comprehensive legislative safeguards. However, there has been a modification in the Brazilian Forest Code (law nº 12.651, dated May 25th, 2012) that has resulted in the reclassification of hypersaline salt-flat zones, or apicum, as areas that are now eligible for development. The reclassification in question has the capacity to make a significant contribution towards the continuous decline of mangrove regions (Ward and Lacerda, 2021). An example of an area that has recently become available for potential development is a hypersaline salt flat, spanning approximately 6,000 square kilometres, situated in the mangrove regions of northeastern Brazil (Ward *et al*., 2023; Ward and De Lacerda, 2021).

Subcriterion A3 measures changes in mangrove area since 1750. Unfortunately, there are no reliable data on the mangrove extent for the entire province during this period, and therefore the Tropical Southwestern Atlantic mangrove ecosystem is classified as **Data Deficient (DD)** for this subcriterion.

Overall, the ecosystem is assessed as **Least Concern (LC)** under criterion A

Criterion B: Restricted Geographic Distribution

Criterion B measures the risk of ecosystem collapse associated with restricted geographical distribution, based on standard metrics (Extent of Occurrence EOO, Area of Occupancy AOO, and Threat-defined locations). These parameters were calculated based on the 2020 Tropical Southwestern Atlantic province mangrove extent (GMW v.3).

Province	Extent of Occurrence EOO (Km^2)	Area of Occupancy (AOO) >1%	Criterion B
The Tropical Southwestern Atlantic	1008490.0	236	**LC**

For 2020, AOO and EOO were measured as 236 grid cells 10 x 10 km and 1008490.0 km^2, respectively (Figure 3). Excluding from the total of 372 those grid cells that contain patches of mangrove forest that account for less than 1% of the grid cell area (< 1 Km^2), the AOO is measured as **236, 10 x 10 km grid cells** (Figure 3, red grids).

Considering the very high number of threat-defined-locations, there is no evidence of plausible catastrophic threats leading to potential disappearance of mangroves across their extent.

As a result, the Tropical Southwestern Atlantic mangrove ecosystem is assessed as **Least Concern (LC)** under criterion B.

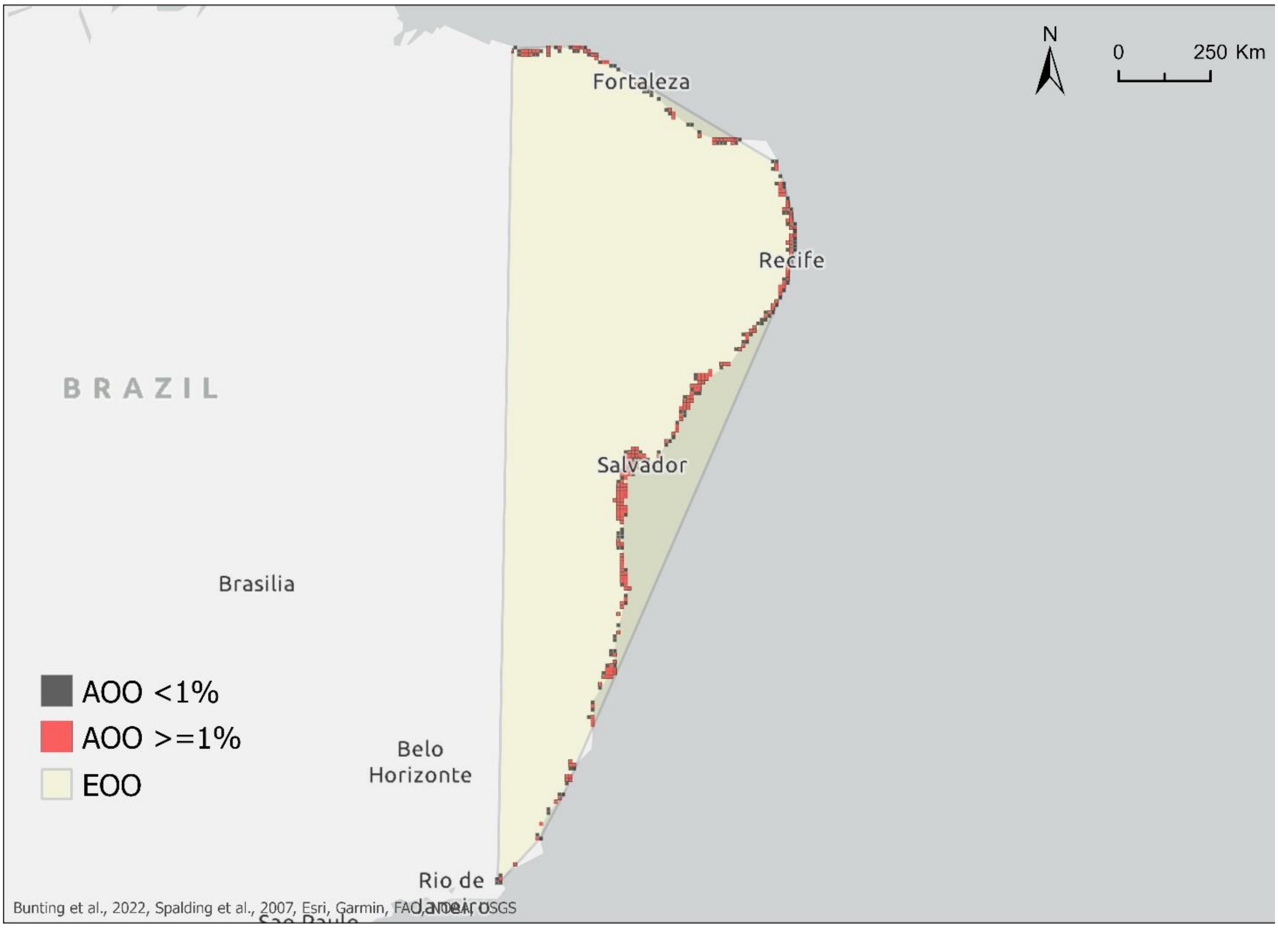

Figure 3. The Tropical Southwestern Atlantic mangrove Extent Of Occurrence (EOO) and Area Of Occupancy (AOO) in 2020. Estimates based on 2020 GMW v3.0 spatial layer (Bunting *et al.*, 2022). The red 10 x 10 km grids (n=236.) are more than 1% covered by the ecosystem, and the black grids <1% (n= 136).

Criterion C: Environmental Degradation

Criterion C measures the environmental degradation of abiotic variables necessary to support the ecosystem. Subcriterion C1 measures environmental degradation over the past 50 years: There are no reliable data to evaluate this subcriterion for the entire province, and therefore the Tropical Southwestern Atlantic mangrove ecosystem is classified as **Data Deficient (DD)** for subcriterion C1.

Subcriterion C2 measures environmental degradation in the future, or over any 50-year period, including from the present. In this context, the impact of future sea level rise (SLR) on mangrove ecosystems was assessed by adopting the methodology presented by Schuerch *et al.*, (2018). The published model was designed to calculate both absolute and relative change in the extent of wetland ecosystems under various regional SLR scenarios (i.e. medium: RCP 4.5 and high: RCP 8.5), with consideration for sediment accretion. Therefore, Schuerch *et al.*, (2018) model was applied to the Tropical Southwestern Atlantic mangrove ecosystem boundary, using the spatial extent in 2010 (Giri *et al.*, (2011) and assuming mangrove landward migration was not possible.

According to the results, under an extreme sea-level rise scenario of a 1.1 m rise by 2100, the projected submerged area is ~ -10.5% by 2060, which remains below the 30% risk threshold. Therefore, considering that no mangrove recruitment can occur in a submerged system (100% relative severity), but that -10.5% of the ecosystem extent will be affected by SLR, the Tropical Southwestern Atlantic mangrove ecosystem is assessed as **Least Concern (LC)** for subcriterion C2.

Subcriterion C3 measures change in abiotic variables since 1750. There is a lack of reliable historic data on environmental degradation covering the entire province, and therefore the Tropical Southwestern Atlantic province is classified as **Data Deficient (DD)** for this subcriterion.

Overall, the ecosystem is assessed as **Least Concern (LC)** under criterion C.

Criterion D: Disruption of biotic processes or interactions

The global mangrove degradation map developed by Worthington and Spalding (2018) was used to assess the level of biotic degradation in the Tropical Southwestern Atlantic province. This map is based on degradation metrics calculated from vegetation indices (NDVI, EVI, SAVI, NDMI) using Landsat time series (≈2000 and 2017). These indices represent vegetation greenness and moisture condition.

Mangrove degradation was calculated at a pixel scale (30m resolution), on areas intersecting with the 2017 mangrove extent map (GMW v2). Mangrove pixels were classified as degraded if two conditions were met: 1) at least 10 out of 12 degradation indices showed a decrease of more than 40% compared to the previous period; and 2) all twelve indices did not recover to within 20% of their pre-2000 value (detailed methods and data are available at: maps.oceanwealth.org/mangrove-restoration/)). The decay in vegetation indices has been used to identify mangrove degradation and abrupt changes, including mangrove die-back events, clear-cutting, fire damage, and logging; as well as to track mangrove regeneration (Lovelock *et al.*, 2017; Santana, 2018; Murray *et al.*, 2020; Aljahdali *et al.*, 2021; Lee *et al.*, 2021). However, it is important to consider that changes observed in the vegetation indices can also be influenced by data artifacts (Akbar *et al.*, 2020). Therefore, a relative severity level of more than 50%, but less than 80%, was assumed.

The results from this analysis show that over a period of 17 years (~2000 to 2017), 2.2% of the Tropical Southwestern Atlantic mangrove area is classified as degraded, resulting in an average annual rate of degradation of 0.13%. Assuming this trend remains constant, +6.6% of the Tropical Southwestern Atlantic mangrove area will be classified as degraded over a 50-year period. Since less than 30% of the ecosystem will meet the category thresholds for criterion D, the Tropical Southwestern Atlantic mangrove province is assessed as **Least Concern** (**LC)** under subcriterion D2b.

No data were found to assess the disruption of biotic processes and degradation over the past 50 years (subcriterion D1) or since 1750 (subcriterion D3). Thus, both subcriteria are classified as **Data Deficient (DD)**.

Overall, the Tropical Southwestern Atlantic ecosystem remains **Least Concern (LC)** under criterion D.

Criterion E: Quantitative Risk

No model was used to quantitatively assess the risk of ecosystem collapse for this ecosystem; hence criterion E was **Not Evaluated (NE)**.

5. Summary of the Assessment

CRITERION			
A. Reduction in Geographic Distribution	A1 Past 50 years	A2 Future or any 50y period	A3 Historical (1750)
	DD	LC	DD
B. Restricted Geo. Distribution	B1 Extent of Occurrence	B2 Area of Occupancy	B3 # Threat-defined Locations > 5
	LC	LC	LC
C. Environmental Degradation	C1 Past 50 years (1970)	C2 Future or any 50y period	C3 Historical (1750)
	DD	LC	DD
D. Disruption of biotic processes	D1 Past 50 years (1970)	D2 Future or Any 50y period	D3 Historical (1750)
	DD	LC	DD
E. Quantitative Risk analysis	NE		
OVERALL RISK CATEGORY	LC		

DD = Data Deficient; LC = Least Concern; NE = Not Evaluated

Overall, the status of the Tropical Southwestern Atlantic mangrove ecosystem is assessed as **Least Concern (LC).**

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7. Appendices

1. List of Key Mangrove Species

List of plant species considered true mangroves according to Red List of Threatened Species (RLTS) spatial data (IUCN, 2022). We included species whose range maps intersected with the boundary of the marine provinces/ecoregions described in the distribution section.

Class	Order	Family	Scientific name[2]	RLTS category
Magnoliopsida	Lamiales	Acanthaceae	*Avicennia germinans*	LC
Magnoliopsida	Lamiales	Acanthaceae	*Avicennia schaueriana*	LC
Magnoliopsida	Malpighiales	Rhizophoraceae	*Rhizophora mangle*	LC
Magnoliopsida	Myrtales	Combretaceae	*Laguncularia racemosa*	LC

[2] According to the RLTS, the true mangrove *Rhizophora racemosa* intersects the northern part of the Trop SW province (Ceará, Brazil region). However, the distribution outlined by Schaeffer-Novelli et al., which is widely accepted, indicates that the ranges of *R. racemosa* and *R. harisonii* extend only up to São Luís in Maranhão state. Nonetheless, it is not certain that *R. racemosa* does not occur in regions beyond this known range, as further studies on the species' distribution, including genetic and taxonomic research, are needed.

2. List of Associated Species

List of taxa that are associated with mangrove habitats in the Red List of Threatened Species (RLTS) database (IUCN, 2022). We included only species with entries for Habitat 1.7: “Forest - Subtropical/Tropical Mangrove Vegetation Above High Tide Level” or Habitat 12.7 for “Marine Intertidal - Mangrove Submerged Roots”, and with suitability recorded as “Suitable”, with “Major Importance” recorded as “Yes”, and any value of seasonality except “Passage”. The common names are shown where available.

Class	Order	Family	Scientific name	RLTS category[2]	Common name
Actinopterygii	Atheriniformes	Atherinopsidae	*Atherinella brasiliensis*	LC	Robust silverside
Actinopterygii	Perciformes	Sciaenidae	*Stellifer brasiliensis*	LC	
Actinopterygii	Syngnathiformes	Syngnathidae	*Hippocampus reidi*	NT	Long-snout seahorse
Aves	Accipitriformes	Accipitridae	*Buteogallus aequinoctialis*	NT	Rufous Crab-hawk
Aves	Caprimulgiformes	Caprimulgidae	*Macropsalis forcipata*	LC	Long-trained nightjar
Aves	Caprimulgiformes	Trochilidae	*Amazilia leucogaster*	LC	Plain-bellied nmerald
Aves	Charadriiformes	Laridae	*Thalasseus acuflavidus*	LC	Cabot's tern
Aves	Charadriiformes	Laridae	*Thalasseus maximus*	LC/	Royal tern
Aves	Coraciiformes	Alcedinidae	*Chloroceryle americana*	LC	Green kingfisher

Aves	Coraciiformes	Alcedinidae	*Megaceryle torquata*	LC	Ringed kingfisher
Aves	Gruiformes	Rallidae	*Aramides mangle*	LC	Little wood-rail
Aves	Gruiformes	Rallidae	*Porphyrio martinica*	LC	Purple gallinule
Aves	Gruiformes	Rallidae	*Rallus longirostris*	LC	Mangrove rail
Aves	Passeriformes	Dendrocolaptidae	*Dendroplex picus*	LC	Straight-billed woodcreeper
Aves	Passeriformes	Furnariidae	*Certhiaxis cinnamomeus*	LC	Yellow-chinned spinetail
Aves	Passeriformes	Passeridae	*Tachycineta albiventer*	LC	White-wingedCrust swallow
Aves	Passeriformes	Thraupidae	*Conirostrum bicolor*	NT	Bicolored conebill
Aves	Passeriformes	Thraupidae	*Sporophila falcirostris*	VU	Temminck's seedeater
Aves	Passeriformes	Thraupidae	*Sporophila frontalis*	VU	Buffy-fronted seedeater
Aves	Passeriformes	Vireonidae	*Hylophilus pectoralis*	LC	Ashy-headed greenlet
Aves	Pelecaniformes	Ardeidae	*Ardea cocoi*	LC	Cocoi heron
Aves	Pelecaniformes	Ardeidae	*Bubulcus ibis*	LC	Cattle egret
Aves	Pelecaniformes	Ardeidae	*Butorides striata*	LC	Striated heron
Aves	Pelecaniformes	Ardeidae	*Egretta caerulea*	LC	Little blue heron
Aves	Pelecaniformes	Ardeidae	*Egretta thula*	LC	Snowy egret
Aves	Pelecaniformes	Ardeidae	*Nyctanassa violacea*	LC	Yellow-crowned Night-heron
Aves	Pelecaniformes	Ardeidae	*Nycticorax nycticorax*	LC	Black-crowned night-heron
Aves	Pelecaniformes	Ardeidae	*Tigrisoma fasciatum*	LC	Fasciated tiger-heron
Aves	Pelecaniformes	Threskiornithidae	*Eudocimus ruber*	LC	Scarlet tbis
Aves	Psittaciformes	Psittacidae	*Primolius maracana*	NT	Blue-winged macaw
Aves	Psittaciformes	Psittacidae	*Touit melanonotus*	VU	Brown-backed parrotlet
Aves	Suliformes	Fregatidae	*Fregata magnificens*	LC	Magnificent frigatebird
Aves	Suliformes	Fregatidae	*Fregatta minor*	LC	
Aves	Suliformes	Fregatidae	*Fregatta trinitales*	LC	Atlantic lesser frigatebird
Aves	Suliformes	Phalacrocoracidae	*Nannopterum brasilianum*	LC	Neotropic cormorant
Aves	Pelecaniformes	Ardeidae	*Cochlearius cochlearius*	LC	Boat-billed heron
Aves	Tinamiformes	Tinamidae	*Crypturellus noctivagus*	NT	Yellow-legged tinamou
Chondrichthyes	Rhinopristiformes	Pristidae	*Pristis pristis*	CR	Largetooth sawfish

Crustacea	Decapoda	Crustacea	*Minuca rapax*	LC	
Crustacea	Decapoda	Ocypodidae	*Leptuca leptodactyla*	LC	
Crustacea	Decapoda	Ocypodidae	*Minuca mordax*	LC	
Crustacea	Decapoda	Ocypodidae	*Uca maracoani*	LC	
Crustacea	Decapoda	Ocypodidae	*Ucides cordatus*	LC	
Crustacea	Decapoda	Gercacinidae	*Cardisoma guanhumi*	LC	
Magnoliopsida	Myrtales	Combretaceae	*Conocarpus erectus*	LC	Silver-leaved buttonwood
Mammalia	Carnivora	Procionidae	*Procyon cancrivorus*	LC	Crab-eating racoon
Mammalia	Cetacea	Delphinidae	*Sotalia guianensis*	NT	Guiana dolphin
Mammalia	Cetacea	Pontoporidae	*Pontoporia blainvillei*	VU	Franciscana
Mammalia	Primates	Atelidae	*Alouatta ululata*	EN	Maranhão Red-handed howler monkey
Mammalia	Primates	Cebidae	*Sapajus xanthosternos*	CR	Buff-headed capuchin
Mammalia	Sirenia	Trichechidae	*Trichechus manatus*	VU	American manatee
Liliopsida	Poales	Poaceae	*Spartina alterniflora*	LC	Atlantic cordgrass
Liliopsida	Alismatales	Cymodoceae	*Halodule wrightii*	LC	Shoal grass
Liliopsida	Alismatales	Hydrocharitaceae	*Halophila decipiens*	LC	Paddle grass
Liliopsida	Alismatales	Hydrocharitaceae	*H. baillonii*	VU	Clover grass
Liliopsida	Alismatales	Ruppiaceae	*Rupia maritima*	LC	Baeked tasselweed
Liliopsida	Poales	Typhaceae	*Typha dominguensis*	LC	Southern cat-tail

3. National Red List of Species

This table include the species assessed or listed in the National Action Plans for the Conservation of Endangered Species in Mangroves (Plano de Ação Nacional para a Conservação das Espécies Ameaçadas e de Importância Socioeconômica do Ecossistema Manguezal; PAN Manguezal). If blank, the species occurs in this province according to the PAN but has not yet been formally assessed.

Class	Order	Family	Scientific name	National/subnational Red List category[2]	Common name
Aves	Charadriiformes	Laridae	*Thalasseus maximus*	EN	Royal tern
Aves	Pelecaniformes	Ardeidae	*Tigrisoma fasciatum*	VU	Fasciated tiger-heron
Crustacea	Decapoda	Ocypodidae	*Leptuca thayeri*		
Crustacea	Decapoda	Ocypodidae	*Minuca victoriana*		
Crustacea	Decapoda	Ocypodidae	*Minuca vocator*		

Crustacea	Decapoda	Ocypodidae	*Minuco burguesi*		
Crustacea	Decapoda	Ocypodidae	*Leptuca cumulanta*		
Crustacea	Decapoda	Gercacinidae	*Cardisoma guanhumi*	CR	
Magnoliopsida	Brasicales	Bataceae	*Batis maritima*		Beachwort
Magnoliopsida	Malvales	Malvaceae	*Talipariti pernambucense*		Embira-do-mangue
Magnoliopsida	Sapindales	Anacardiaceae	*Schinus terenbenthifolius*		Brazilian pepper tree
Liliopsida	Poales	Poaceae	*Spartina densiflora*		Dense-flowered cordgrass
Liliopsida	Alismatales	Cymodoceae	*H. emarginata*		Shoal grass

4. National Data for Subcriterion A

Compilation of existing historical data on the extent of mangroves in the Trop Southwest Atlantic Province.

Country	Year	Mangrove Area (Ha)	Reference
Brazil	1972 to 1975	1,380,000	Herz R. (1991). Manguezais do Brasil. São Paulo. Instituto Oceanográfico, Universidade de São Paulo. 227 pp.
Espírito Santo state, Brazil	1970	13,690	IEMA, Instituto Estadual de Meio Ambiente e Recursos Hídricos. (2023). Mapeamento e análise espaço-temporal dos manguezais do Espírito Santo: Comparativo entre 1970, 2007 e 2015. Ed. Roberto Hezer Moreira Vervloet. Accessed in 04/01/2024: https://iema.es.gov.br/cgeo
Espírito Santo state, Brazil	2015	11,417	IEMA, Instituto Estadual de Meio Ambiente e Recursos Hídricos. (2023). Mapeamento e análise espaço-temporal dos manguezais do Espírito Santo: Comparativo entre 1970, 2007 e 2015. Ed. Roberto Hezer Moreira Vervloet. Accessed in 04/01/2024: https://iema.es.gov.br/cgeo
Brazil	1985	1,010,263	Diniz, *et al*. (2019). Brazilian Mangrove Status: Three Decades of Satellite Data Analysis. Remote Sensing, 11(7), 808. https://doi.org/10.3390/rs11070808 MAPBIOMAS Collection v.8.0 (https://brasil.mapbiomas.org/)
Brazil	1996	1,034,414	Diniz, *et al*. (2019). Brazilian Mangrove Status: Three Decades of Satellite Data Analysis. Remote Sensing, 11(7), 808. https://doi.org/10.3390/rs11070808 MAPBIOMAS Collection v.8.0 (https://brasil.mapbiomas.org/)
Brazil	2020	1,038,359	Diniz, *et al*. (2019). Brazilian Mangrove Status: Three Decades of Satellite Data Analysis. Remote Sensing, 11(7), 808. https://doi.org/10.3390/rs11070808 MAPBIOMAS Collection v.8.0 (https://brasil.mapbiomas.org/)
Brazil	2020	1,141,470	Bunting, *et al*. (2022). Global Mangrove Extent Change 1996–2020: Global Mangrove Watch Version 3.0. Remote Sensing, 14(15). doi:10.3390/rs14153657
Brazil	1996	1,147,456	Bunting, *et al*. (2022). Global Mangrove Extent Change 1996–2020: Global Mangrove Watch Version 3.0. Remote Sensing, 14(15). doi:10.3390/rs14153657
Brazil	1999 to 2003	1,300,000	Spalding M, Kainuma M, Collins L (2010). World Atlas of Mangroves (version 3.1). A collaborative project of ITTO, ISME, FAO, UNEP-WCMC, UNESCO-MAB, UNU-INWEH and TNC. London (UK): Earthscan, London. 319 pp. DOI: https://doi.org/10.34892/w2ew-m835