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MANGROVES IN THE ANTHROPOCENE: CLIMATE CHANGE AND CONSERVATION CHALLENGES

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Aerial roots of *Rhizophora stylosa* in Fiji by Shigeyuki Baba

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Sonneratia alba dominant forest in Thailand by Shigeyuki Baba

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MANGROVES IN THE ANTHROPOCENE: CLIMATE CHANGE AND CONSERVATION CHALLENGES

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Foreword

I am glad that the International Society for Mangrove Ecosystems (ISME) will be publishing the ISME Mangrove Ecosystems Technical Reports, Volume 7, on Mangroves in the Anthropocene: Climate Change and Conservation Challenges. This technical report is prepared in collaboration with the International Institute for Environmental Studies (IIES) at Trent University in Peterborough, Ontario, Canada, and with the Federal University of Ceará in Fortaleza, Brazil. Partnership between ISME and IIES is in the pipeline and this will be beneficial for both parties.

ISME is an international, non-profit and non-governmental scientific society, established in August 1990 and is based in Okinawa, Japan. Stipulated in the statutes of the society, ISME shall collect, evaluate and disseminate information on mangrove ecosystems, and shall promote international cooperation. The three main foci of ISME are the application of mangrove knowledge; mangrove training and education; and exchange of mangrove information. Activities of ISME include collaborations with organizations, universities, research institutes, and local communities. As of February 2025, the membership of ISME stands at more than 1,300 individual members and 49 institutional members from 94 countries/regions.

This volume consists of nine chapters written by mangrove experts from six countries. Covering important topics on mangrove ecosystems in relation to climate change, carbon sink, blue carbon and conservation challenges, I hope these chapters that are available online and in hard-copies will benefit government departments, NGOs, societies, and students from universities and colleges.

Prof Emeritus Dr Sanit Aksornkoae

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Preface

While mangrove forests account for less than one percent of tropical forests by land area, they have a disproportionate impact on coastal communities and coastal ecology. As brackish wetlands, they have a complex ecology that supports a wide array of ecosystem services and provide important economic benefits to local communities. Mangrove ecosystems are characterized by high biodiversity, intensive cycling of nutrients within the systems and unique abilities of adjusting to constantly changing physical-chemical conditions.

While mangrove ecosystems have a highly efficient and stable ecology when not subjected to anthropogenic pressures, they can succumb to a range of human influences. These include deforestation, over-utilization of biota, changes in the hydrologic conditions of the forests related to rising sea levels and/or reduced freshwater inputs due to up-stream dams, and temperature changes associated with climate change, to name a few. As a result, mangrove deforestation is a growing problem in many parts of the globe. Quite correctly, many have referred to the mangrove ecosystem as a ‘canary in the coal mine’ for climate change.

The potential of using mangrove forests to capture atmospheric carbon release, i.e. blue carbon, is immense, as much as 40 times that of a tropical rainforest on a per unit area by some estimates. But many factors can affect this potential, including anthropogenic drivers such as sea level rise, changing atmospheric temperatures, damming of up-stream waterways, etc. While the potential for reducing atmospheric carbon using mangrove forests is real, there are still many aspects of mangrove ecology, particularly those related to stressors, that remain to be studied and understood.

The chapters in this volume cover a range of topics relevant to the preservation and management of mangrove forests, as well as the extension of them to support local communities and to act as a carbon sink. The authors present important background data on the extent of mangrove forest degradation, the roles of mangroves in nutrient and contaminant cycling, strategies for the management of mangroves by local communities (e.g., repopulation of degraded environments), and the expansion of mangrove forests for the capture of blue carbon. The economic value of mangroves is also explored. The volume concludes with an assessment of future prospects for the precious mangrove resources as well as some of the knowledge gaps that lead to uncertainty in management practices.

The International Institute for Environmental Studies (IIES) is honoured to be a sponsor of this important series of technical papers. Chapters in this volume originated as lectures, organized by Professor Dr Lacerda, represent one of the IIES on-line lecture series. Recorded lectures are available at https://ii-es.com/events/mangroves-in-the-anthropocene-climate-change-and-conservation-challenges of our website.

Professor Dr Douglas Evans

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The collection of papers presented in this book, on the challenges faced by mangroves in the Anthropocene, results from the initiative of the International Institute of Environmental Studies (IIES), Trent University, Canada that promoted a series of lectures on this important topic. I would like to thank Prof Douglas Evans, Director of the IIES, for his strong support of the seminar series of this book, and Ms Raquel Scott for her technical support. I am grateful to Prof Huy Dang, who introduced me to Trent University that made this international cooperation possible.

I thank the International Society for Mangrove Ecosystems (ISME), Okinawa, Japan, my mangrove home more than 30 years ago, for adopting this book project; in particular Prof Dr Shigeyuki Baba, the ISME Executive Director and my life-long friend and collaborator, and Dr Hung Tuck Chan, the ISME Publications Editor. My sincere gratitude to all the chapter authors, and mangrove scientists who have collaborated with me on many different projects. I would like to express my deep gratitude to Dr Hayla E. Evans, who has kindly offered to review the first draft of the chapters and ended up improving them considerably.

I sincerely thank the Conselho Nacional de Desenvolvimento Científico e Tecnológico (CNPq), and the Fundação Cearense de Apoio ao Desenvolvimento Científico e Tecnológico (FUNCAP), Brazil. The financial support of the International Institute of Science and Technology of Continent - Ocean Materials Transfer (INCT-TMC Ocean) has enabled my stay at Trent and in the making of this book.

Professor Dr Luiz Drude de Lacerda

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Chapter 1

Mangrove forests: A summary of their biogeography, distribution and biology, and challenges they face in the Anthropocene

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1. Biogeography

1.1. A summary of mangrove paleobiogeography

Mangroves are trees, shrubs and ferns uniquely well-defined in space with intertidal boundaries marked by high and low tide levels (Vannucci, 2001) that grow in tropical and subtropical coastal zones. Today they occur along the shorelines of 123 countries and territories and cover an estimated area of about 147,256 km^2 (~15 million ha) (Leal and Spalding, 2024). They represent less than 1% of tropical forests worldwide and less than 0.4% of the global total forest area. However, their unique ecology renders them a huge environmental and economic importance globally, including an extremely efficient mitigation tool to face climate change impacts, mostly as a barrier to extreme climatic and oceanographic events and a long-term sink for carbon dioxide and other greenhouse gases (Alongi, 2014).

There is agreement based on sound fossil records that higher plants adopted the living in intertidal environments in the Tethys Sea from the Late Cretaceous through the Early Tertiary between 100 and 80 My, when the fossil record of most mangrove genera, as well as many genera of gastropods associated with mangrove forests appeared (Duke 1995; Allison et al., 1999). By the Palaeocene (~60 My), pollen of the mangrove palm *Nypa* (currently the only palm tree found solely in mangroves), *Rhizophora* and *Avicennia* were already abundant in Late Tertiary deposits through the Americas. Although a higher number of taxa is found in Southeast Asia, all lines of evidence support a vicariance hypothesis based on nested distribution patterns at the community and species-level suggesting three independent regions of diversification of mangrove ecosystems, namely, Southeast Asia, the Caribbean and the Eastern Pacific-Indian Ocean region (Allison et al., 1999; Lacerda, 2002).

The mangrove palm *Nypa* is the earliest of the modern mangrove genera to appear in the fossil record (Rull, 1998). Late tertiary-known fossil fruits and pollen of *Nypa fruticans* are identical to the fruits and seeds of Cretaceous and Palaeocene *Nypa* and from the Mid- to Late-Eocene records, when *Nypa* abounded in South America, Africa and Australia. For example, in the Maastrichtian of Cameroon, *Nypa*-type pollen comprises up to 20-25% of all pollen preserved in the samples (Salard-Cheboldae, 1981), suggesting widespread swamp development.

The mangrove flora in the Americas is believed to have evolved much later. The genera *Rhizophora* and *Avicennia*, in particular, may have travelled through the Tethys Sea into what is presently the Mediterranean Sea. From there they migrate to the Atlantic coast of the Americas and to West Africa, reaching the Caribbean by the early Eocene (some 55-50 My), when the distance from the Tethys Sea was shortest and travel was made favourable by a warm interval during the Cenozoic (Haq, 1981). Palaeoceanographic current patterns during the Palaeocene, when the Tethys Current dominated regional surface circulation, support this scenario of mangrove migration (Haq, 1984). Prior to the late Eocene, pollen of *Brevitricolpites variabilis* and *Brevitricolpites*-like palynomorphs were among the most abundant types. These plants dominated the same habitat now occupied by *Rhizophora*, suggesting a clear equivalency between the two genera. However, *Brevitricolpites* eventually disappeared from the palynofloras in the middle Eocene (Graham, 1995). After the isolation of the American mangroves, the endemic genus *Laguncularia* became established throughout the Atlantic shore of the continent. Differential extinction at the generic level (e.g., *Nypa* in the Americas) occurred throughout the pan-tropical distribution of mangroves, following regional and local environmental changes; also, drastic reduction of some genera (ending-up in present-day relics), restricted small populations, e.g., *Pelliciera rhizophorae* to the Caribbean region.

1.2. Present global distribution

Presently, mangroves have a pan-tropical distribution, including most oceanic islands, except some Pacific islands encompassed by meridians 90º and 170º, where mangroves do not exist; this suggests an unlikely trans-Pacific migration route as proposed by some authors (Woodroffe, 1987; Tomlinson, 1994). A total of 17 plant families and approximately 70 taxa of true (exclusive to mangroves) and numerous associated taxa (frequent in mangroves) are found in mangrove forests. Most (~60) taxa of trees occur in Asia and Australasia, with only about 10 in the Americas. Approximately 16% (11 taxa) are endangered or threatened with extinction. The total mangrove forests area globally is ~15 million ha; the largest tracts are in Asia (40.4%), the Americas, including the Caribbean (27.6%), Africa (20.8%), and Australia and New Zealand (7%), with the remaining 4.1% is distributed among the Pacific Islands (3.9%) and the Middle East (0.2%) (Leal and Spalding, 2024). It is estimated that mangrove forests provide at least US $1.6 billion each year in ecosystem services and support a large number of coastal livelihoods worldwide (Ferreira et al., 2022). Mangroves' latitudinal limits are set by minimum winter temperatures, in most cases, or harsh environmental circumstances, such as the extreme desert conditions along the Pacific Coast of South America and Southern Western Africa. Along the American Pacific coast, mangroves extend from latitude 30°15′ N, in Puerto de Lobos, Mexico, to latitude 5°32′ S only, at the Piura River estuary in northern Peru, because of cold water upwelling from La Niña causes extremely arid conditions south of this location. As well, a lack of suitable landforms, and extraordinary fluctuations in river flow, particularly during El Niño events, make the establishment of permanent mangrove seedlings, difficult (Cintrón-Molero and Schaeffer-Novelli, 1992). On the Atlantic coast, mangroves extend northward to Bermuda (Latitude 32°20′ N) and to Laguna, in South Brazil (Latitude 28°30′ S).

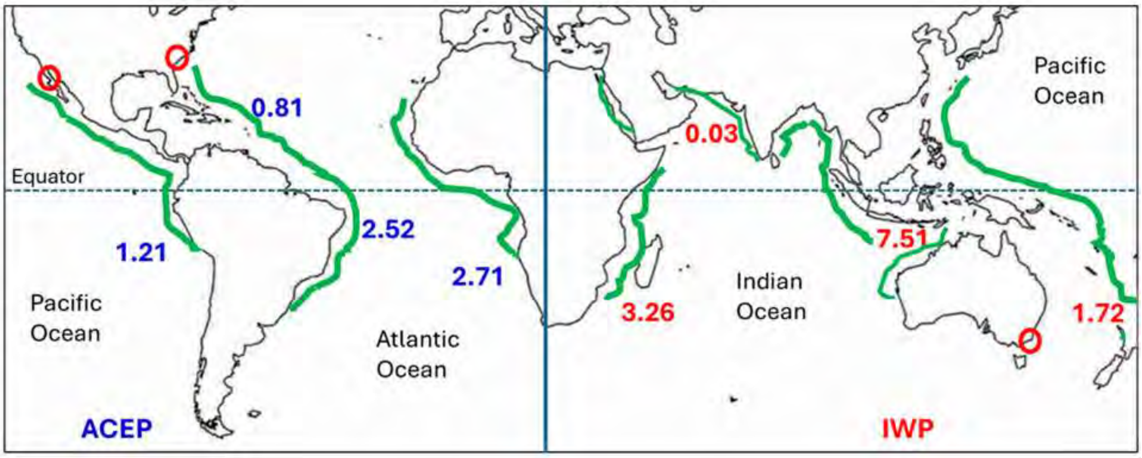


Figure 1. Worldwide distribution and approximate mangrove cover in different geographical areas: ACEP = Atlantic, Caribbean & East Pacific (7.3 Gha); IWP: Indo-West Pacific (12.5 Gha). Largest mangrove areas by country: Indonesia (4.5 Gha); Australia (1.3 Gha); Brazil (1.2 Gha); Nigeria (1.0 Gha). Open red circles are latitudinal limits presently changing at rates higher than 1° degree due to global warming.

Along West Africa, mangroves occur from Mauritania in the north (Iles Tidra, 19°50’ N) to Angola in the south (Rio Longo estuary, 10°18′ S) (Spalding et al. 2010), whereas in East Africa, mangroves are distributed from the Gulf of Aden (Zeylac and Lowya Cadde, Somalia at 11° 43′ N) to the Nahoon River at 33° S in South Africa (Naidoo, 2023; SIMAD, 2023).

The Indo-West Pacific, including Australasia, harbours the most biodiverse mangrove forests with about 70 true mangrove species, including a large number of economically important trees, shrubs and palms. This region also encompasses the largest contiguous deltaic mangroves worldwide at the Ganga-Brahmaputra, the Irrawaddy and the Mekong River deltas. Mangroves extend from Tanegashima Island, Kyushu, Japan, (Lat. 30°34′26″N) to Westernport Bay, Australia (Lat. 38°21′ S).

Although this region is mostly humid tropical, it includes three main arid coastal zones; i). the coastal zone bordering the western and southern coasts of Australia and the Indian Ocean sub-region; ii). The Arabian Sea, including the Gulf of Aden, the Red Sea, the Gulf of Oman; and iii). The coasts of Baluchistan and Sindh in Pakistan and of Gujarat in India, where oceanographic conditions coupled with climate and air-sea interface phenomena create an especially arid climate with low precipitation (Vannucci, 2001; Saenger et al., 2019).

1.3. Human-induced changes in mangrove biogeography

Human activities are promoting significant changes in mangrove distribution and biogeography worldwide. Changes include within-region distribution of indigenous species as well as invasion of exotic ones. Drivers are either intentional or non-intentional actions on local, regional and global scales. The introduction of tree species of particular economic importance is a more common driver of changing mangrove species biogeography. Re-introduction of the mangrove palm, *Nypa fruticans*, in Central America and West Africa is a typical example of intentional changes. Although pollen of *Nypa* was abundant in the Tertiary fossil record in the Americas, it was eradicated from this continent by the Earlier Holocene, following an increase in aridity (Duke, 1991). Similarly, the same climate change event restricted *Pelliciera rhizophorae* (another abundant species in the region), to small populations in northern South and Central America (Castillo-Cárdenas et al., 2005). Alternatively, a recent example of non-intentional introduction is the first record of the flowering *Sonneratia apetala*, a pan-Asian mangrove tree recently reported in Cubatão mangrove in Southeastern Brazil, which is close to Santos Harbour, the largest port in South America. This specific location strongly suggests that this exotic species invaded and became established in the local mangroves, most probably because of release of ballast water from the intense international maritime traffic in the region and the proximity of the local mangroves to port facilities (Eysink et al., 2023).

A large-scale phenomenon associated with climate change is the shift in the latitudinal limits of mangroves worldwide, as minimum winter temperatures rise. Faizlioglu et al. (2020) and Chen et al. (2021) reviewed present-day latitudinal limits of mangrove species throughout the tropics. They reported poleward migration of a dozen mangrove species in all continents. Notable are those species where latitudinal shifts larger than one degree. These are *Avicennia germinans* and *Rhizophora mangle* in the North Atlantic coast of America and *Rhizophora mangle* and *Laguncularia racemosa* in the North Pacific coast. On the Pacific coast of Australia, *Aegiceras corniculatum* and *Avicennia marina* also migrated poleward by 1.2° and 2.4°, respectively (Faizlioglu et al., 2020).

Also associated with climate change is the landward migration of mangroves, particularly in semi-arid coastal areas, due to decreasing freshwater supply from continental watersheds and increasing sea level. In Northeastern Brazil, most of the seaward fringe of mangroves shows lower sediment accretion rates than the local average sea level rise (Ward et al., 2023). Many estuaries are witnessing extensive mangrove expansion landward, increasing their overall cover by 24% region-wide, but reaching over 400% in some cases (Godoy and Lacerda, 2015; Godoy et al., 2018).

Local anthropogenic activities, mostly associated with aquaculture, rice fields, oil palm plantations, urban expansion and draining (among others), change mangrove coverage worldwide, including localized species extinctions. On the other hand, reforestation and afforestation projects, both through direct planting/seeding and through hydrological restoration and control of disturbances that favour natural regeneration, have expanded mangrove area worldwide (Contessa et al., 2023). Globally, the highest proportion (40%) of threatened mangrove species is found along the Atlantic and Pacific coasts of Central America, where about 10% of true mangrove species are listed as *Near Threatened* and another 30% are listed as *Vulnerable*; mostly these are rare or uncommon species endemic to this region, such as *Pelliciera rhizophorae*. Worldwide, between 20% and 35% of mangrove area has been lost since 1980 at an annual rate of about 0.5%, with local rates estimated as high as 2-8%. These rates are as high as or higher than those in upland tropical wet forests. Due to their strategic location, it is possible that in 1980, total mangrove area represented 50% to 70% of its original cover prior to European colonization, when port cities were established by European settlers in support of intercontinental navigation.

Table 1. Estimated mangrove loss (ha) due to human pressure on mangroves in the major distribution regions: ACEP = Atlantic, Caribbean & East Pacific; IWP: Indo-West Pacific. Data is based on original estimates from Bunting et al. (2022) and Contessa et al. (2023).

Region	1996	2020	% change
ACEP	66,555	64,920	↓ 2.5
IWP	86,024	74,797	↓13.1
Total	152,579	139,717	↓ 8.25

At the global scale, Contessa et al. (2023) estimated a total mangrove area of about 14.8 million ha (±1.4%) in 2020 and that the rate of annual mangrove loss had decreased from 2000 to 2010 relative to 2010 to 2020, largely because of decreasing rates of loss in South and Southeast Asia and the expansion of reforestation.

2. A summary of mangrove biology and ecology

2.1. Forest types

Mangrove forest structure is initially driven by propagule establishment, which depends on the interaction of several abiotic (climate, geomorphic settings, tides) and biotic (soil invertebrate and microorganism diversity, herbivory, bioturbation) factors (Ferreira et al., 2019). Once established, forest architecture, biomass and structural complexity reflect the broad spatial variation of the environmental conditions within a given site, e.g. inundation frequency and duration (linked to geomorphology, hydrodynamic conditions and tides), salinity, freshwater and nutrient availability, which are the key parameters influencing forest structure; therefore, structural characteristics of mangrove forests are frequently site specific.

A summary of major physical and structural characteristics of mangrove forests is presented in Figure 2 (adapted from Lacerda et al., 2022). Four general types of forests can be identified, Riverine; Basin; Fringe and Overwash, although local variability may be high. This division is mostly based on their specific location but eventually results in different functional and structural attributes. Riverine mangroves occur along estuaries extending landward and, depending on the extent of saline intrusion, can develop into extensive and broad stretches of mangroves with complex forest structure and high biomass, particularly in most humid areas and in large deltas. In more arid conditions or close to their latitudinal limits, riverine stands occur in narrow fringes along rivers and estuary margins, typically having lower canopy density and overall biomass that are frequently eroded as trees are blown down by the wind.

Basin forests typically occupy the landward portion of mangroves, where flooding occurs mostly during spring tides leading to high soil salinity due to strong evaporation. These extreme conditions result in high organic matter accumulation in the soils, but with a poorly structured, stunted vegetation, frequently dominated by the salt excreting *Avicennia* species.

Well-developed fringe forests occur along the broad mudflat expanses typical of coasts adjacent to large rivers, which deposit enormous amounts of continental sediments, for example, the littoral zone adjacent to the Amazon River estuary. These broad mudflats provide protection from high-energy marine influences. Under drier climates, ocean forcing, waves and currents, as well as stronger winds, readily prevent any pioneer fringe forests from establishing, restricting them to the relatively protected waters of estuarine mouths within the littoral fringes of coastal lagoons. Overwash forests occur in recently formed fluvial and estuarine islands that generally contain medium to coarse sands (Lacerda et al., 2007; Godoy and Lacerda, 2014). There, they are flooded daily by tides with slow accumulation of organic matter in soils, most of the litterfall being exported to adjacent waters. This forest type is sometimes also seen colonizing beach rock outcrops at the intertidal level.

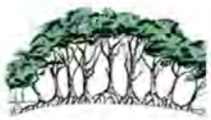
Type	Sediment origin	Geomorphology	Forest architecture	General configuration
Riverine	**Terrigenous** from fluvial transport, clastic & siliciclastic sediments	Low energy open coast mud flats Estuarine & Deltaic	Tall (> 10 m) trees with high biomass. Freshwater wetland plants mix at the upper estuary	
Basin	**Terrigenous** from fluvial transport clastic & **Marine** from tidal flow, **organo-clastic particles**	Lowlands behind riverbanks and fringing forest	Varying in height (3-10 m) with intermediate to low biomass. Tidal dominated	
		Salt flats behind or within mangrove forests	Stunted "dwarf" trees (<3 m), high soil salinity.	
Fringe	Mostly remobilized, **marine clastic** sediments from rivers, **carbonates and bioclastic sands**	Tidally dominated small river estuaries	Varying in size, intermediate to high biomass	
		Lagoonal (with bioclastic sediments)	Low biomas	
Overwash	**Marine, carbonates, bioclastic sands**	Open waters	High energy, freshwater scarcity	

Figure 2. A summary of major mangrove forest types, geomorphology, sediment sources and major forest structural characteristics.

2.2. Biological challenges in a stressful environment

Mangroves are a gift of the tides (cited from Vannucci, 1989). To live by the tides, however, is not easy! They must develop numerous adaptations to live in a humidity and salinity-variable, muddy, frequently anoxic habitat while also being exposed to extreme sea events. Notwithstanding, among the high diversity of trees that evolved into mangroves, there is a convergence in adaptations to these different drivers that induced similar morphological and physiological features. These include characteristic branching and rooting systems composed of aerial and cable roots and pneumatophores, that anchor the plant into a semi-fluid substratum and increase tree resistance to wind and sea events; also special reproductive strategies, such as viviparity, and seeds and propagules able to survive in sea water for long periods, allow long-range dispersal by currents and a continuous gene pool flux among mangroves. Most of the dispersal routes discussed above results from these properties.

Adjusting to soil and pore waters having highly variable salinity and anoxia, requires specialized physiological mechanisms. A thickened exodermis as well as extensive and well-developed aerenchyma enable mangrove species to guarantee the necessary oxygen supply to the submerged roots in a suboxic or anoxic environment. Among other common mechanisms to cope with stressful soil conditions are selective, 'filtering', i.e., differential absorption or rejection of various salts and other compounds (typical of *Rhizophora*) and salt exclusion and excretion glands at the root or leaf levels (as in *Avicennia*), or by immobilization in hypersaline vacuoles or special storage tissues (Lacerda et al., 2024). To cope with low oxygen at the root level, mangroves have lenticels, i.e., tissue pores that allow air to pass through the spongy tissue (aerenchyma) to the roots. They also are capable of strong radial oxygen liberation (ROL) oxidizing their rhizosphere and precipitating iron oxides as iron plaques at the external root surface. ROL helps the plants avoid toxic by-products of anoxia (e.g. sulphides) by oxidizing them back to sulphate, whereas iron plaques strongly bind heavy metals in porewaters as non-bioavailable chemical species. Mangroves are considered to be sources and/or sinks of chalcophiles and other trace metals that precipitate and accumulate in sediments as sulphides, a product of the sulphate reduction metabolism typical of these ecosystems. These mechanisms have been thoroughly studied and are dealt with in the specialized literature and elsewhere.

3. Threats to mangrove ecosystems

The major drivers threatening mangroves changed in variability intensity and speed over the past 40 years depending on the region. Drivers of impacts in the second half of the 20th century, such as coal and salt production, are today minor and site-specific. On the other hand, shrimp aquaculture, which was relatively restricted during that period, has now expanded to be a global and major threat to mangroves worldwide. A comprehensive regional assessment of mangroves published in the early 1990s, details the state of mangroves during the second half of the 20th century with respect to major drivers, pressures, state changes, impacts and the resulting societal response, and provides an overview of their sustainable utilization and conservation status (ISME-ITTO, 1993). Many review papers summarized major present-day drivers of environmental impacts on mangroves (e.g., Lacerda et al., 2019; Friess et al., 2020; Bunting et al., 2022; Ferreira et al., 2022; Contessa et al., 2023; Naidoo, 2023).

3.1. Drivers of direct impacts on mangroves

During the last 40 years of the 20th century, rapid socio-economic growth occurred in most tropical countries. At that time, two major drivers were responsible for the strongest worldwide environmental pressures on mangroves, i.e., urbanization and industrialization (Table 2). These drivers led to intense conflicts regarding mangrove conservation that resulted in increasing non-sustainable uses of mangrove products. Both drivers were strongly associated with extensive internal migrations of populations to the coastline and the need for job creation, land reclamation and water and food supply for growing urban settlements. This triggered the over-exploitation of many mangrove products (wood for fuel, timber and associated fisheries) by traditional populations responding to a fast-growing demand. Also, the increase in coastal population density occurred much faster than the capacity for building the necessary infrastructure, resulting in inadequate disposal of urban solid wastes, wastewaters, sewage and industry wastes, as well as the encroachment of mangroves within large metropolitan areas. This led to rapid forest degradation and loss, even in countries where mangroves were under legal environmental protection.

Although urbanization and industrialization remain significant sources of impact on mangroves in some regions, some optimism arises from an observed decrease in mangrove loss rates following growing worldwide conservation efforts and stronger environmental regulations (Friess et al., 2020). An example is the impact of oil spills, which have decreased drastically in the 21st century, relative to the last three decades of the past century, despite an overall increase in the oil and gas industry. The total area of mangroves killed by oil decreased to almost zero in some regions such as in Latin America and the Caribbean (Duke, 2016). This resulted from strengthened regulations, preparedness and mitigation measures adopted by the industry and government.

Habitat deterioration of peri urban mangrove areas is, however, still a major threat to mangroves in many countries, mostly due to effluents containing toxic substances and excess nutrients that are released from generally unregulated urban and industrial development. In addition, inappropriate disposal of solid wastes, mostly plastics and wastewaters, still triggers localized pollution and eutrophication thus increasing the pressure on the biological resources of mangroves (Lacerda et al., 2019).

Table 2. Summary of drivers of direct impact, pressures and impacts on mangroves and their variation from the late 20th century and today. Trend: refers to importance and intensity, based on expert analysis of the original information.

Drivers	**Main pressures**	**Main impacts**	**Response**	**Trend (1970-2000)**	**Trend (present)**
Urbanization and tourism	Solid waste and wastewater disposal; conversion	Contamination; eutrophication; mangrove eradication	Coastal zone management plans; improving waste treatment; including mangroves into urban green architecture	**Major** Widespread	**Major to Intermediate** Localized
Industry	Effluents disposal; oil spills	Contamination of the biota; tree and fauna mortality	Stronger regulations; improving waste treatment and technology; banning tank washing; improving security and contention measures	**Major** Restricted to industrialized nations	**Intermediate to Minor** Restricted to industrialized nations
Shrimp farming	Conversion; nutrient fluxes	Deforestation; eutrophication	Initial regulation laws; public awareness	**Intermediate to Major** Localized	**Major** Widespread
Forestry	Wood and wood products	Deforestation	Restraining mangrove wood use; extractive reserves; community-based management	**Minor** Localized	**Minor** Localized
Salt production	Conversion	Deforestation	Abandoning ponds	**Minor** Localized	**Minor** Localized

Despite strengthened environmental regulations and technologies controlling effluent discharges, as of 2010, a significant proportion of mangrove species is threatened. In Latin America and the Caribbean region, for example, 4 of the 10 mangrove species present along the Pacific coast of Central America are considered 'Threatened' and one, *Rhizophora samoensis* is listed as 'Near threatened'. Three of the region's mangrove species, *Avicennia bicolor*, *Mora oleifera* and *Tabebuia palustris*, rare species restricted to the Pacific coast are listed as ‘Vulnerable’ (Polidoro et al., 2010).

Minor drivers of direct impacts on mangroves in the late 20th century were forestry, restricted to a few sites in Southeast Asia and Central America and salt production in the localized semi-arid coast of Africa, Latin America and the Caribbean, causing localized deforestation and mangrove conversion. Today, however, countless previously-converted mangrove areas are subject to growing initiatives of mangrove rehabilitation, including natural and assisted replanting programs (Ferreira et al., 2023). In the 20th century, shrimp aquaculture was restricted to certain countries such as the southern Guayas Province, Ecuador, where pond construction is estimated to have converted over 300,000 ha of lush mangroves; shrimp aquaculture only also was increasing its economic importance in a few countries in Asia (ISME-ITTO, 1993; Lacerda, 2002; Lacerda et al., 2019). In the first decade of the present century, however, shrimp aquaculture skyrocketed with a 20% per year growth rate worldwide and becoming the most significant source of direct impacts on mangroves today (Lacerda et al., 2021). Intensive shrimp farming causes immediate deforestation, even in countries where farms cannot be established directly in mangroves, shrimp aquaculture increases nutrient fluxes (causing eutrophication), augments sediment and particle fluxes causing higher siltation and in extreme cases, mangrove burial. Building ponds also change local hydrology, which impacts mangroves by reducing dilution of sea water and freshwater supply, directly affecting mangrove functioning and some ecosystem services, such as nutrient retention and productivity, carbon sinks, and the protection of coastlines (Marins et al., 2020). Initially, regulation laws could not face the increasing economic importance of shrimp farming and so most countries legally released new areas for pond construction. Also, as in most key food and income-generating activities, capital pressures, the equivocate use of 'sustainability' policies and insufficient public awareness as to the importance of mangrove conservation vis-à-vis aquaculture, resulted in failed efforts at conservation and management in many regions (Ferreira et al 2016; Ferreira and Lacerda, 2022).

Direct impacts cause effects easily noticed by society (e.g. mangrove fauna and flora immersed in oil, urban waste accumulation in mangrove forests, etc.), thereby triggering immediate societal responses, such as strengthening existing control policies, augmenting conservation efforts in the creation of new conservations sites (e.g. RAMSAR, MPA, National Mangrove Forests, Extensive Restoration and Replanting Programs, etc.) and improving conservation and management strategies. This societal response was able to control major drivers, decreasing or even reverting their impacts on mangrove ecosystems (Friess et al., 2020).

3.2. Drivers of indirect impacts on mangroves

Drivers of indirect impacts on mangroves and their variability over time are presented in Table 3. Today, drivers of indirect impacts are much more likely to affect mangroves than those causing direct impacts on mangrove ecosystems. Also, contrary to direct impacts, their pressure is difficult to control or minimize by existing national legislations. Frequently, they need multi-lateral agreements and international laws to make their control feasible. Also, their effects are cumulative and chronic, hampering rapid identification at an early stage and hence triggering the necessary societal responses (Lewis et al., 2016).

The damming of rivers was a significant threat to mangroves in the late 20th century. Damming remains and important driver of indirect impacts on mangrove forests today, especially in semi-arid coasts, where water withdrawn to support the activities of a growing population along river basins, may increase its effects. The regulation of river fluxes by dams decreases the contribution of freshwater and continental sediments to the coast, affecting sediment and salt balance and nutrient fluxes, causing erosion of fringe forests and increasing sedimentation in basin forests. Augmenting sea water intrusion increases soil and pore water salinity with a direct impact on mangrove growth and productivity (Lacerda and Marins, 2002; Zhang et al., 2022; Suwarno et al., 2014).

The first quarter of the 21st century witnessed accelerating global climate change, with global temperature reaching maxima record almost every year. Contrary to most expectations raised by the Kyoto Protocol and the many UN Conference of the Parties (COP) meetings held in the past 30 years, greenhouse gas (GHG) emissions and their concentrations in the atmosphere are still on the rise. Among the many threats associated with global warming, those with particular significance to mangrove ecosystems are sea level rise and the increasing frequency of extreme climatic events. Although global in occurrence, global warming impacts are unevenly distributed and interact with regional drivers, modulating the severity of the effects on mangrove ecosystems. Similarly, mangrove responses also vary according to forest typology and ecosystem functioning (Alongi, 2015; Ward et al., 2016).

Sea level rise, a global effect of climate change, is the best studied pressure on mangroves, because dated sediment cores ranging in time from decades to millennia provide insightful records of mangrove response to this pressure. Also, the availability of tidal gauges and the recent rapid development of remote sensing techniques, facilitate its monitoring. Sea level rise increases salinity of surface and ground waters and, under certain circumstances, such as during an extremely dry season or under semi-arid climates, salinity can surpass mangrove regulating capacity by means of filtration and/or excretion, negatively affecting plant growth and biomass accumulation (Ward et al., 2016). A general global warming-associated decrease in annual rainfall observed in many coastal areas, exacerbates the impacts of sea level rise (Godoy and Lacerda, 2015; Ward et al., 2023). Saline intrusion can decrease functional diversity and carbon storage of the forest by affecting less saline-resistant, higher carbon content species as has been observed in the species-rich Sundarbans mangroves, for example (Rahman et al., 2021). Similar results have been reported in other regions in Southeast Asia (Komiyama et al., 2019).

Table 3. Summary of drivers of indirect impact, pressures and impacts on mangroves and their variation from the late 20th century and today. Trend refers to importance and intensity, based on expert analysis of the original information.

Drivers	Main Pressures	Main Impacts	Response	Trend (1970-2000)	Trend (Present)
Damming	Sediment and salt balance; nutrient fluxes	Erosion of coastal forests; siltation; increasing soil and pore water salinity	Watershed committees including coastal communities' representatives	**Major** Mostly in semi-arid regions	**Major** Mostly in semi-arid regions
Climate change	Sea level rise; increased salinity; decreased freshwater supply; extreme events; global warming	Deforestation; erosion; decreasing productivity; landward migration; poleward migration	Poleward and inland migration; GHG abatement	**Minor** Not yet fully recognized	**Major** Global
Agriculture	Nutrient and chemical effluents; land reclamation	Eutrophication; contamination of the biota; deforestation	Watershed communities regulating land uses; restriction on agrochemicals use	**Intermediate** Most from large-scale intensive agriculture	**Intermediate to Minor** Most from large -scale intensive agriculture
Fisheries	Fisheries products	Overfishing and decreasing stocks	Community-based management; establishing temporal closures	**Minor** Widespread	**Intermediate** Widespread and strengthening to due decreasing stocks

Landward migration is one of the most documented responses of mangroves to sea level rise. Where coastal geomorphology allows, such as in coastal plain littoral zones, mangroves migrate upstream in estuaries following the saline intrusion; however, many regions' mangroves may be limited landward by geomorphologies that impede their migration, such as the granitic mountains of southeaster Brazil or the sabkha (salt flat) environments along the coast of the Arabian Gulf and the Red Sea, among others (Godoy and Lacerda, 2015).

A rise in sea level causes erosion, especially in fringe mangroves, accelerating organic matter oxidation and GHG emission. Oxidation of sulphides and the dissociation of metal-sulphides, release associated chalcophile metals, thus facilitating the formation of metal-chloride and metal-organic complexes, and increasing metal bioavailability. This results in positive feedback to global warming (by decreasing mangrove capacity to act as a long-term sink for carbon) and may result in metal pollution and contamination of biological resources. Predictions regarding the decline of the carbon stock in above-ground and below-ground biomass and soil carbon content under future climate change scenarios, as well as events pertaining to heavy metal contamination, are matters of concern, but still far from predictable (Lacerda et al., 2020).

The increasing frequency of extreme events is a verified planetary response to global warming and their impacts on mangroves have been studied in particularly sensitive areas, such as the Caribbean and the monsoon dominated coasts of Southeast Asia. The mangrove forest devastated by typhoon Yolanda in November 2013 in the Philippines, affected the rate of regeneration of trees and natural regeneration of seedlings. An inventory of the number of dead trees, length and number of regenerated shoots, and the number and height of seedlings showed that regeneration differed significantly, depending on the species and the specific geomorphology of sites. For example, the area covered by *Sonneratia* species varied from 18 to 40%, and *Avicennia* species from 19 to 25%, whereas *Xylocarpus* species varied from 50 to 88%. The natural regeneration of seedlings also varied, being highest in *Avicennia* species (73%), followed by *Bruguiera* species (71%), with *Rhizophora* species and *Ceriops* species showing the lowest regeneration percentage (42%) (Alura and Ortega, 2016).

The 2004 Indian Ocean tsunami that caused the death of over one quarter million people and immense economic loss, demonstrated the capacity of preserved mangroves to attenuate its impact and increased public awareness as to the important ecosystem service of mangroves (Dahdouh-Guebas et al., 2005). In addition, wave reduction was similar among different trees compositions, suggesting the efficacy of both natural and replanted forests (Hashim and Catherine, 2013). Wave attenuation by mangroves is effective against all marine events, and high stem density seems to contribute to resistance to storms (Peereman et al., 2022).

The increase in mean air temperature has led to a poleward migration of mangrove geographical distribution limits, following an increase in minimum winter temperatures at the latitudinal limits of mangroves. Mangroves are expanding their latitudinal range mostly on the Atlantic USA coast (Zomlefer et al., 2006) and Australia's east coast. This expansion corresponds to the poleward extension of mild temperature zones during the past half century (Canavaugh et al., 2014). However, this poleward movement of mangroves does not occur on a global scale. Latitudinal expansions are inhibited by barriers to dispersal in some locations, e.g., *Avicennia marina* (Forsk.) Vierh. in Australia and New Zealand (de Lange and de Lange 1994) and Southeast Africa (Adams and Rajkaran, 2020); for unknown reasons, poleward migration is not occurring at other latitudinal limits, such as Western South Atlantic (Soares et al., 2012). To what extent poleward migration will be hampered by a mix of local environmental variables and settings is still far from understood as are any effects on mangroves beyond their colder distribution limits.

Global climate change is intensifying and the impacts on mangroves are only preliminarily being evaluated. It is not the objective of this chapter to extensively review the literature on the subject. Many recent comprehensive reviews on what we know about this subject are available and readers can refer to UUGS (2004); Alongi (2015); Godoy and Lacerda (2015); Ward et al. (2016); Jennerjahn et al. (2017) and Moomaw et al. (2018).

Two major drivers of indirect impacts on mangroves have remained or increased in importance in the present century. Relative to the 20th century, agriculture practices have evolved to become intensive, fertilizer-dependent and commodity-based industries. This has resulted in increasing loads of sediments, nutrients and pesticides, that eventually reach the coast where they can accumulate. Eutrophication and contamination of natural resources are the most environmentally significant outcomes, normally having delayed effects that are not easily associated with this driver. The specific impacts on mangroves are difficult to access, but some studies have demonstrated changes in nutrient cycling efficiency due to excess inputs from anthropogenic sources (Feller et al., 2014; Lovelock et al., 2009; Fauzi et al., 2014; Marins et al., 2020).

Fisheries have no direct impact on mangroves, provided the activity is practised by traditional fishers, and are, as a rule, sustainable. However, the increase in large fishing fleets and fishing gear, has resulted in large disposal of plastics into the oceans, with some eventually ending up along mangrove dominated shores. Today, fishing gear makes up 50-100% of plastic debris found in parts of the ocean, and this may be an underestimate (Apete et al., 2024). More recently, the accelerated increase in aquaculture raised interest (from commercial fisheries) in the use of by-catch products as fish meal for aquafeeds; however, this practice impacts the recruiting of juvenile fish that eventually use mangroves and potentially cause other unknown impacts on mangrove food chains.

4. Concluding remarks

Recent intensification of existing anthropogenic drivers and the emergence of new ones suggest the need for long-term efforts to understand their impacts on mangroves. Most of what we know of mangrove responses to challenges presented by regional and/or global environmental changes is still mostly restricted to mangrove biological components, leaving a large gap in our understanding of mangroves geochemical and biogeochemical responses, and their ecosystem services as a buffer for estuarine pollution and as a carbon sink. Processes considering the continuum between watersheds and the ocean in mangrove-dominated areas are poorly comprehended due to the near absence of long-term monitoring efforts covering large geographical areas. The few existing studies are frequently threatened by economic instability of many nations along the world's tropics.

The current situation calls for joint efforts to promote multi-disciplinary research encompassing the mosaic of natural and impacted mangrove ecosystems worldwide and linking watershed processes with the coastal receptor. Research should be directed towards a comprehensive view of alterations in the fluxes, transformation and the cycle of substances at the continent-ocean interface where, mangroves thrive, and with an emphasis on pollution, eutrophication, oxygen depletion, and any consequent biological crises. Understanding the implications of these changes to ecosystem functioning, services, conservation and sustainable management is urgent, as is the vulnerability of mangrove-dominated regions and the respective threats to society.

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Chapter 2

Mangrove ecology and functioning: The role of biotic functional groups on mangrove functionality and challenges to climate change

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1. Mangroves biology and morpho-physiological features

Mangroves are one of the most striking coastal forests of tropical and subtropical regions. They live in the frontier between water and land of estuarine, deltaic and sheltered coasts. Mangroves are wetlands and as such, water is the primary controller of their physical-chemical environment and the ecology of their biota. These forests exist mainly in two large biogeographical realms, the Atlantic-Caribbean-East Pacific (ACEP) and the Indo-West Pacific (IWP). The IWP was the centre of mangrove origin and dispersion in the Eocene period (Ricklefs and Latham, 1993), and has more species than ACEP. In fact, the species in the ACEP (except two neotropical endemic genera, *Laguncularia* and *Pelliciera*), are a subset of IWP genera that arrived to new world from the Eocene. The IWP mangroves include the genera *Heritiera, Aegiceras, Excoecaria, Sonneratia, Lumnitzera, Bruguiera, Ceriops, Kandelia, Xylocarpus, Rhizophora* and *Avicennia*, the two latter being shared with ACEP stands (Table 1). Not only mangrove species are more abundant in the IWP, other taxonomic groups that include flowering plants, algae, polychaete and non-polychaete worms, crustaceans, molluscs and fishes are also more diverse in the IWP than in the ACEP (Ricklefs and Latham, 1993).

Table 1. Mangrove regions (for updating see Contessa et al. 2023; FAO, 2023), number of genera and exclusive species in the ACEP and the IWP biogeographical regions (modified from Ricklefs and Latham, 1993); * if a new species of *Pelliciera* is validated (Duke, 2020).

Taxonomic Diversity of Mangrove Taxa in Different Biogeographical Regions			
Region and sub-region	**Area of mangrove habitat (km^2)**	**Number of genera**	**Exclusive species**
IWP			
2. Australia/New Guinea	17,000	16	35
1. Asia/Indonesia	52,000	17	39
6. East Africa/Madagascar	5,000	8	9
ACEP			
5. West Africa	27,000	3	5
4. Western Atlantic/Caribbean	48,000	3	6
3. Eastern Pacific	19,000	4	8*

Mangroves are facultative halophytic plants, that is, they can live in tidal marine flooded areas, with several species able to live in brackish-fresh water. Because they are intertidal colonizers, i.e., they live in the dynamic space between low and high tides, these forests are a link between the marine and the terrestrial environments. The intertidal area is a harsh environment in which to live, because organisms are subjected to air, intense sunlight and varying salinity, which pose a high thermohaline stress. Mangroves live in warm temperatures (min. 20°C and less than 5°C variation), and need in general calm environments to establish and develop. This is due to their characteristic viviparous buoyant seeds, called propagules. They can float and remain viable for weeks or months in the sea until they root, furnishing a great capacity for ecosystem expansion. These forests occur in diverse climatic regions, from humid with high annual rainfall, where they can reach huge sizes (e.g. in Central-West Africa (Gabon, Equatorial Guinea) and the Pacific coast of Colombia), to dry areas such as the Middle East, the Caribbean Sea and other semi-arid coasts. There are also tall forests stands in Amazonia and in some areas of Southeast Asia.

Among the most interesting features of mangroves are their roots, that allow trees to stand in soft, muddy waterlogged soils (Figure 1). These roots promote high sedimentation rates, and the muddy sediment layer can reach meters deep. However, the sediment is anoxic, because the high primary production from mangroves (litter fall) is degraded by countless populations of microorganisms, mainly bacteria, that rapidly exhaust oxygen from the waterlogged soil to oxidize the deposited organic matter (OM). As a result, mangroves have evolved physiological mechanisms to take oxygen directly from air. For example, some mangroves (*Rhizophora* species, *Avicennia* species and *Bruguiera* species) have specialized tissues, 'lenticels' (Figure 1j) in their aerial roots and pneumatophores, to absorb oxygen directly from the air. In the root zone (i.e., 10 cm diameter around root) the organic matter and nitrite/nitrate content are higher, while pH and C:N ratio are lower, suggesting high bacterial N assimilation and retention close to the roots (Inoue et al., 2011). Mangroves show diverse types of roots: stilt/prop roots (*Rhizophora*), buttress roots (*Heritiera*), pencil roots (*Avicennia*), cone roots (*Sonneratia*), knee roots (*Bruguiera*) and spreading roots (*Ceriops*) that allow their development in soft, muddy soils (Takarina, 2019).

Another feature that allows mangroves to live in the harsh conditions of high salinity (called an 'osmotic desert') is their capacity to concentrate salts in their tissues to equalize external osmotic pressure and avoid being dehydrated by water loss. Because sodium chloride (NaCl) is problematic for live tissues, some mangroves (*Avicennia* and *Aegiceras*) also excrete salts by special glands in their leaves, as well as in roots and pneumatophores. *Rhizophora* root lenticels, for example, contribute to salt excretion (Bento et al., 2024). Other mangroves filter salt by through their roots or concentrate salt in disposable tissues like old leaves or bark. Mangroves concentrate other substances (of high molecular weight), for example, tannins. Tannins are phenolic compounds that bind to proteins, cellulose and starches, creating insoluble substances that are resistant to decomposition and a deterrent of herbivory; as well as they protect the leaves from the intense sunlight.

The habitat of mangroves is constantly changing. This mean that mangrove forests have a great heterogeneity at both temporal and spatial scales, or dimensions. This is due to abiotic (coastal geomorphology, rainfall, tidal amplitude, temperature, salinity and soil characteristics like nutrients, oxygen content, grain size and humidity) (Krauss et al., 2008; Lacerda et al., 2022), and also biotic (soil micro- and macro-organism community composition, bioturbation, pollination and herbivory (Cannicci et al., 2008; Kristensen, 2008; Ferreira et al., 2015, 2019a) factors. In general, ecosystem structural heterogeneity in general leads to a more diversified biota, because more resources (habitats, food, shelter, etc) lead to more niches that can be home to more species; this is true also for mangrove ecosystems, especially for the extremely diverse IWP forests (Ferreira et al., 2024). Tree species determine the forest structure and architecture, which influence forest productivity and so their biomass, along with other important features, such as the physical-chemical features of the sediments, biotic diversity, resistance and resilience to natural and human degradation drivers, among others.

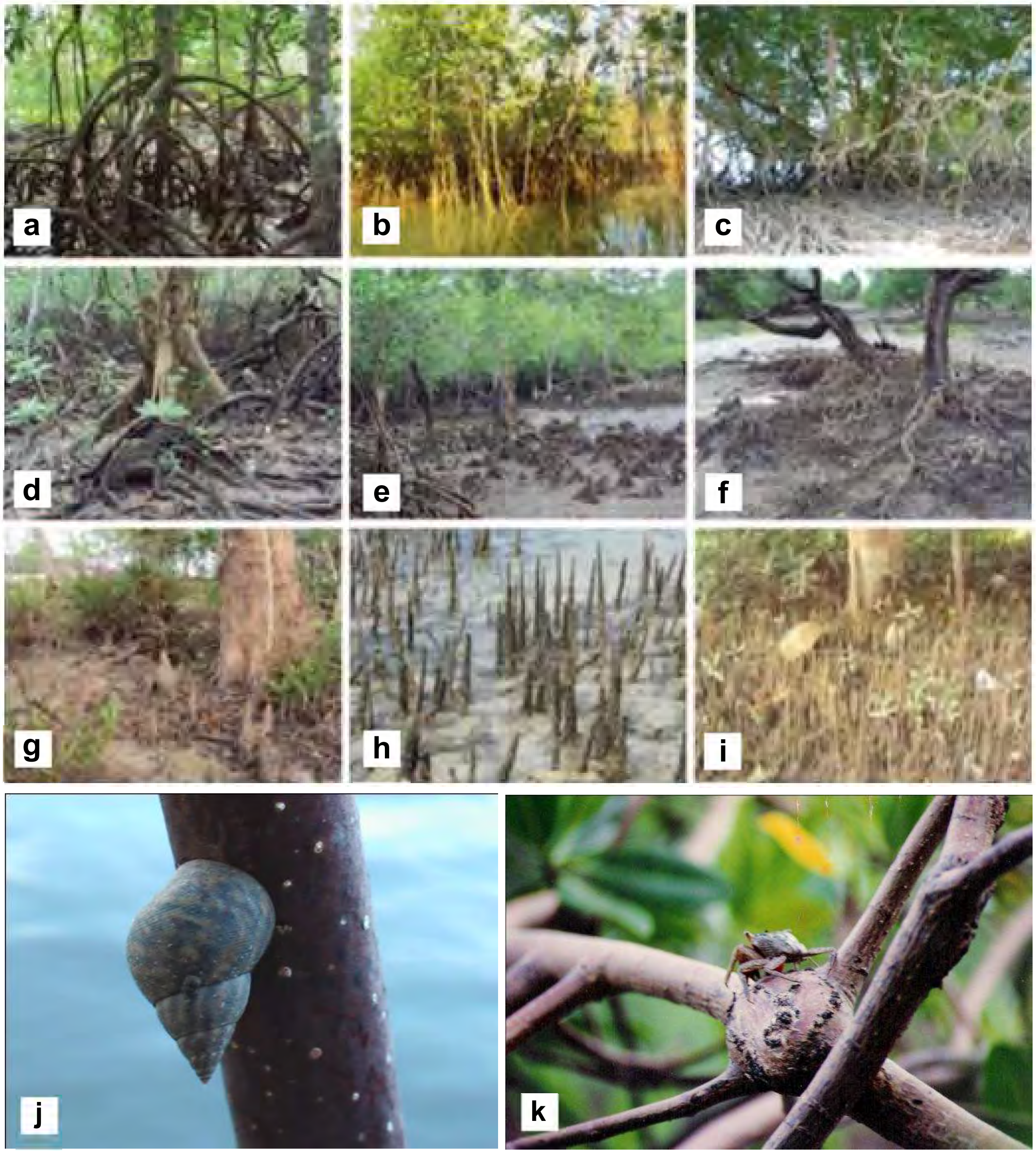

Figure 1. Different mangrove root systems: a. *Rhizophora* species; b. Stilt root, c. Prop roots; d. Plank roots of *Xylocarpus*; e. Knee roots of *Bruguiera* species; f. Above ground roots of *Lumnitzera* species; g. *Xylocarpus moluccensis*; h. Pneumatophores of *Sonneratia*; i. *Avicennia* (modified from Saxena et al., 2013); j. Lenticels (white points) of and *Littoraria* species attached to a *Rhizophora* prop root; k. Several macrobenthic invertebrates live in the root realm, with different species occupying different tree heights (which may change as they age, like the pictured *Aratus pisonii*), from burrows surrounding roots to canopy.

2. Mangrove productivity, nutrient cycles and ecological roles

Mangroves are one of the most productive terrestrial ecosystems, and their high productivity comes from photosynthetic rates that sequester high amounts of carbon from the atmosphere and fix it as vegetal organic matter. This primary production goes to the soil mostly as litter fall. There, a huge pool of different microorganisms (bacteria, archaea, protists, fungi and algae) in the soil, decompose particles of this material and transform them into dissolved OM. This detritus-microorganisms complex is the major energy source for detritus-consumer organisms, as well as for detritus-consumer organisms. Therefore, detritus fuels the mangrove food chain (Figure 2).

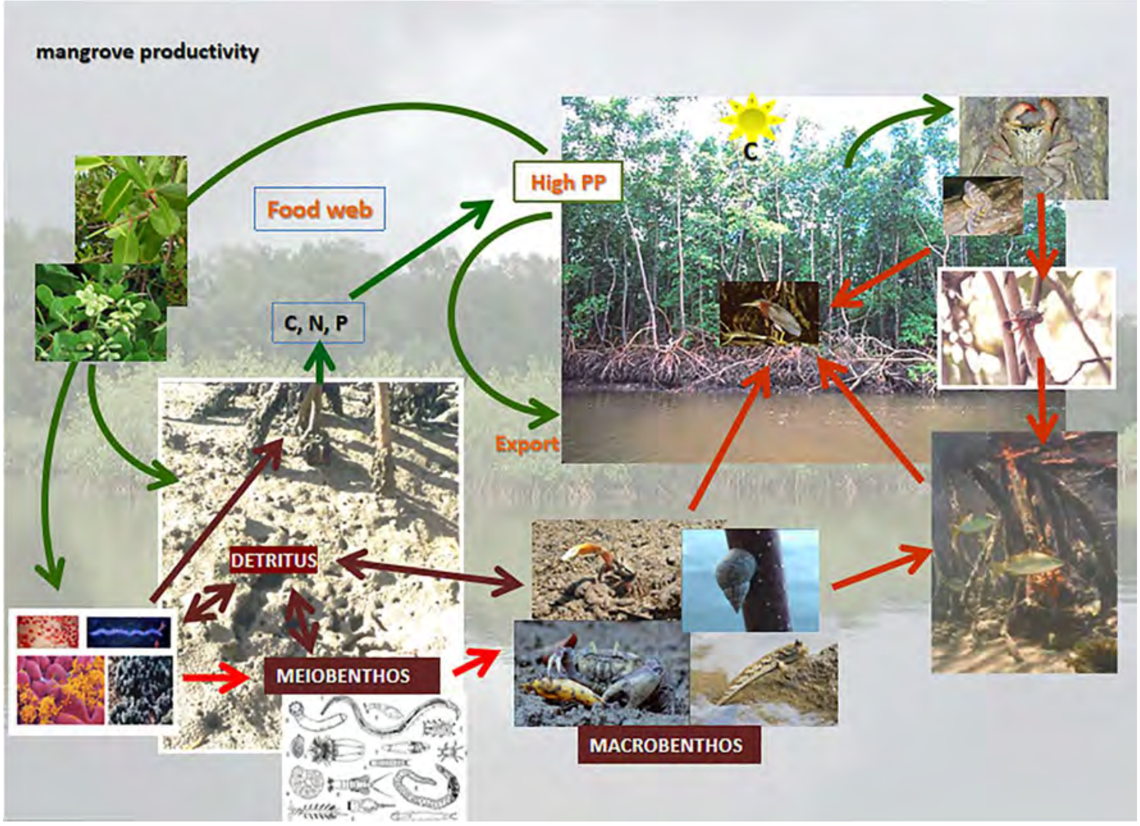

Figure 2. Mangrove food web. Setae indicate resource flow; bidirectional setae show detritus production and consumption. Green setae are vegetal matter/nutrient paths; red setae indicate trophic paths; black and brown setae show detritus production/consumption paths.

Similar to microorganisms, a pool of countless benthic invertebrates contributes to OM cycling, by chopping litter into particles and/or by consuming/producing detritus. Macrobenthic organisms are conspicuous inhabitants of mangrove ecosystems, for example, several species of crustaceans (cirripeds, shrimps, crabs and isopods) and molluscs (gastropods, teredinids and bivalves). Some of them (i.e., sesarmid crabs and gastropods) are fully or partially arboreal. Most macrobenthic invertebrates and a large part of the microbiota are endemic to mangrove forests, and some key species, or groups of species, are extremely important for mangrove ecosystem core functioning (Ferreira et al., 2024).

For example, several chemical processes associated with mangrove growth (and so forest structure) depend on the microbiotic composition of the sediment (Holguín et al., 2001; El-Tarabily et al., 2021; Farrer et al., 2022). Crabs and shrimps, and some juvenile and adult fish species, excavate and maintain burrows in the soil for refuge (Barletta et al., 2000; Kristensen, 2008; Ferreira et al., 2019b; Lira et al., 2021), influencing sediment microbiota, structure and physical-chemical conditions (redox potential in particular) (Lacerda et al., 1993), which have implications for root and stem development and hence tree growth. Herbivores (and omnivores) also play significant roles in the mangrove forest structure through their trophic function, for example by consuming specific tree species (Lacerda et al., 1986) and plant propagules and favouring the predominance of other trees or feeding at multiple trophic levels (Ferreira et al, 2015).

Groups of organisms (from same or different taxonomic groups) that perform the same ecological role, or function are examples of functional groups (FGs). Mediators of soil biogeochemistry (which include decomposers), soil bioturbators/burrowers, herbivores/omnivores, wood-borers and pollinators are some FGs involved in these key processes in mangroves (Cannicci et al, 2021;

Ferreira et al, 2024) (Figure 3). There are other FGs in mangroves, depending on the scope and ecological variables used in the studies characterizing them, but those discussed here, in particular, are crucial to mangrove functioning and physical structure.

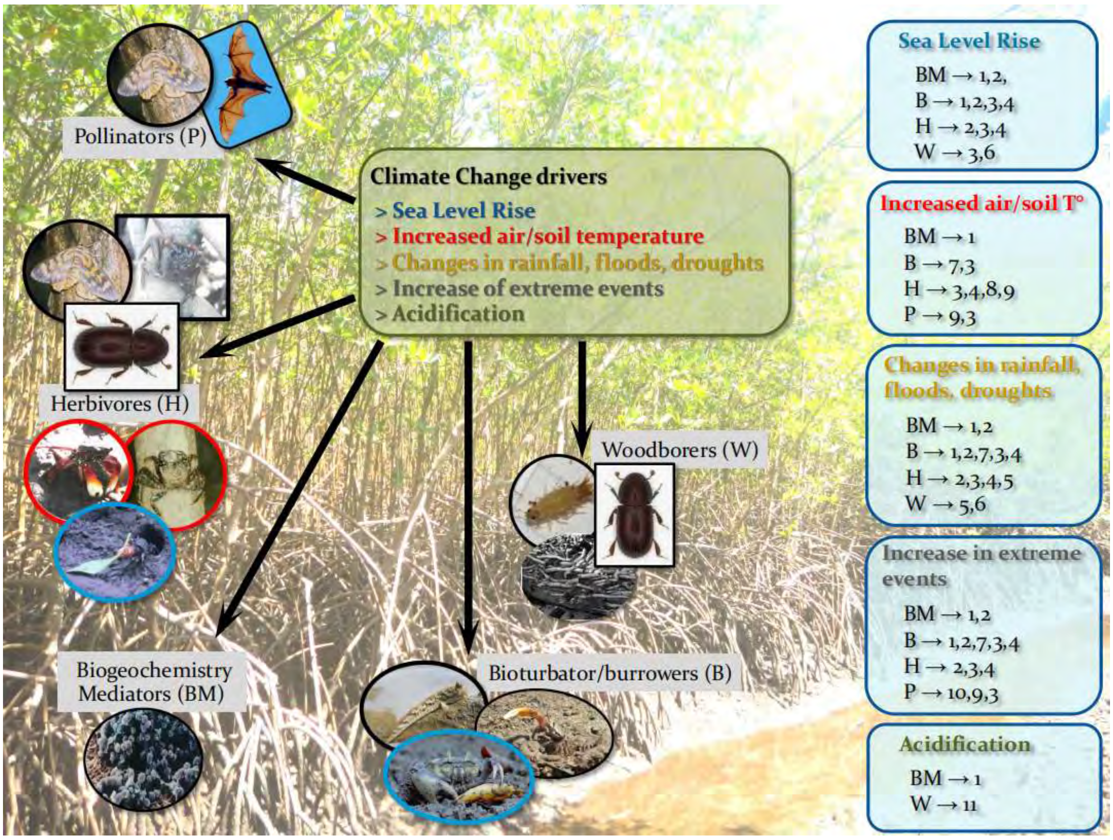

Figure 3. Effects of climate change (in green box) on functional groups (FGs) and the direct effects of FGs on the forest (blue boxes) (the numbers of the effects do not necessarily express event order): (1) Disruption to soil biogeochemical processes; (2) Decreased nutrient availability > impact on forest productivity; (3) Changing forest structure and biomass/C stock; (4) Changes in propagule recruitment patterns; (5) Changing existing forest zonation patterns; (6) Decrease in forest structural resistance; (7) Decrease/increase sediment aeration by sediment reworking; (8) Mass defoliation; (9) Disruption to tree development; (10) Disruption to pollination and reproductive output; (11) Decrease in inputs of OM, litter and deadwood processing, and nutrient cycling reduction. [*Note:* The herbivore FG includes the several mobile Grapsoids (several sesarmids in the IWP, and *Goniopsis cruentata* (Grapsidae) and a few sesarmids in the ACEP), that live in forest soil and climb trunks and roots, also with omnivore and preying habits (in red circles). Some of these crabs are simultaneously bioturbator/burrowers and major herbivores (in blue circles)] (modified from Ferreira et al., 2024).

Soil microorganisms are extremely significant and crucial for mangrove ecology and functioning, since they are responsible for the cycling of nutrients (C, N, P and S) and other elements (e.g., metals) in mangrove soils (Holguín et al., 2001; Alongi, 2021; El-Tarabily et al., 2021). Organic matter is aerobically decomposed by fungi and bacteria producing carbon dioxide. In the anoxic deeper sediment layers, anaerobic decomposition of OM by dissimilatory sulphate reduction performed by bacteria and archaea, produces methane and sulphides. Microphytobenthic organisms, such as Diatomacea and Cyanobacteria contribute by burying carbon and trace elements (Brown et al., 2021; Lacerda et al., 2022). On the other hand, N-fixing bacteria extract this element from the main pool where it exists (the atmosphere), and transform it into forms bioavailable to plants (i.e. ammonia), through cycles of nitrification and denitrification, producing N oxides and elemental N again (Holguín et al., 2001; Alongi, 2021). At the same time, mycorrhizal fungi and phosphate solubilizing bacteria release phosphorus attached to humic compounds and make it bioavailable to plants and the biota. This is an extremely important function for mangrove nutrition. Sulphate reducing bacteria are the most significant OM decomposers in mangrove anaerobic soils, mineralizing organic matter and making N, iron (Fe) and P more available for the biota (Holguín et al., 2001; Abhijith et al., 2018; Alongi, 2021).

Semi-terrestrial brachyuran crabs (mainly grapsoids, ocypodoids and some xanthoids), are one of the most ecologically significant key groups, performing several crucial roles in mangroves (Ferreira et al., 2024). Herbivore crabs, through the consumption or burial of specific mangrove propagules, determine tree species that develop and hence forest richness, architecture and biomass. Consumption of mangrove leaves by crabs (*Ucides, Sesarma, Aratus* and *Goniopsis* in ACEP, and *Clistocoeloma*, *Episesarma*, *Neosarmatium*, *Parasesarma* and *Parasesarma* in IWP) is the main shredding process of vegetal matter transformation into detritus. Litter can decompose for many weeks in burrows while being consumed by crabs, leading to a significant decrease in tannin content and an increase in N content through microbial metabolism, which improves the nutritional value of the leaf litter (Ashton, 2002). Studies suggest that a decrease in populations of folivore crabs (*Ucides occidentalis*) (along with a longer inundation time) can lead to higher OM exportation and lower transference rate of nutrients to higher trophic levels, further decreasing secondary production and biomass (Castellanos-Galindo et al., 2017).

However, most mangrove crabs excavate individual or collective burrows to protect themselves from thermohaline stress, allowing oxygen penetration in the sediment thus increasing the aerobic degradation of OM. These ecological roles make them significant ‘ecosystem engineers’ (Smith et al., 1989; McKee, 1995; Bosire et al., 2005; Ferreira et al., 2013, 2015) because they directly or indirectly control the availability of resources to other organisms by promoting physical state changes in biotic or abiotic materials (Jones et al., 1994). Physical ecosystem engineering by organisms is the physical modification, maintenance or creation of habitats. In the IWP mangroves, there are fishes that perform a similar function. Noteworthily, higher aerobic microbiota metabolism inside burrows releases carbon dioxide (CO_2), which is less heat-retaining greenhouse gas (GHG) than methane (CH_4) produced by anaerobic metabolism.

In addition to herbivore crabs and gastropods (Figure 1j,k), insects can be significant in vegetal matter consumption, and also in pollination. In the ACEP stands, trees are mostly pollinized by wind and some non-specific pollinators; however, this functional group of insects is more significant in the IWP, where approximately half of more than 3,000 species of insects found in the mangroves are mangrove specific (Yeo et al., 2021). Wood-borer insects and molluscs contribute to fragmenting high molecular weight vegetal tissues and can erode tree structure if infestation is increased by higher temperatures or extended floodings (Hendy et al., 2022; Ferreira et al., 2024).

Mangrove fauna also can include countless vertebrate species that inhabit or occasionally visit mangroves, such as fishes, birds, amphibians, reptiles, mammals and others, several of them depending on mangroves as feeding and breeding grounds (see review by Arceo-Carranza et al., 2021).

3. Potential disruption of ecological roles of FGs by climate change

While the direct effects of climate change (CC) on mangroves are being extensively studied (Ward et al., 2016; Ward and Lacerda, 2021; Alongi, 2022), the indirect effects of CC through impacts on the associated biota are still poorly understood (Ferreira et al., 2023). Biotic components and FGs are directly affected by forest degradation, from human causes or by the effects of CC, currently one of the most significant threats to mangroves. CC can affect the crucial ecological roles of FGs that directly ‘shape’ the structural (and hence architectural) features of the forest (tree species, density, size and biomass). This can threaten structural/physical resistance and the resilience of mangroves to further CC-driven pressures such as sea level rise (SLR), increasing storminess, and extreme floods/droughts, hastening functional alterations (sometimes in cascade) and forest degradation (Ferreira et al, 2024). Figure 3 shows the predicted effects of CC on FGs and the direct effects of FGs on the forest.

Recent studies show that mangrove invertebrate macrofauna (at least crustaceans and molluscs) have extremely low functional redundancy (i.e. every species performs ecological function(s) in part, or entirely different from the ecological function(s) of others) relative to intertidal position, diet and behavior (Ashton et al., 2003; Cannicci et al., 2021; Delfan et al., 2021). Extremely low functional redundancy of ecological functions performed by only one species is common in 60% of the 16 worldwide mangrove locations studied, showing the vulnerability of these forests to species loss; models predict that systems with low functional redundancy are more prone to experience functional loss and species extinction (Henderson et al., 2020; Cannicci et al., 2021).

4. Concluding remarks

Mangrove forests are life-support ecosystems that are extremely important for the ecosystemic goods and services they provide, for humans and for countless coastal species of microorganisms, invertebrates and vertebrate organisms. The role and importance of biota in the conservation of mangroves and in their provision of goods and services is increasingly recognized and need to be addressed, so that species and their ecological roles can be protected. With respect to the CC emergency, mangroves greatly contribute to the mitigation several direct and indirect effects of changes in climate patterns over coastal areas, sometimes at their own expense. Thus, the conservation and protection of extant forests, and the rehabilitation/restoration of degraded ones is of paramount importance.

Several rules for reforestation also apply to mangrove ecosystems to maximize biodiversity recovery. 1) protect existing pristine forest to conserve functional diversity, 2) use natural regeneration when possible, 3) maximize biotic recovery to meet multiple goals including climate change mitigation and 4) select appropriate areas and species for restoration to maximize mangrove biodiversity and function. To fully apply these rules, however, further research is needed regarding key FGs and their multi-trophic biotic interactions in mangrove ecosystems, so that responses to CC and the wider implications on sustainable livelihoods and food security services that mangroves provide can be understood. To effectively face CC challenges, rehabilitation/restoration of mangrove forests with respect to their functional processes is crucial. Efforts should first concentrate on restoring/recruiting key FGs that recover several important ecological processes for the community including enhanced species richness and functionality.

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Chapter 3

Mangrove forests: Impacts of climate change and mitigation potential

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1. Impacts of climate change on mangrove ecosystems

Mangroves occur along protected and/or sheltered coastlines throughout the tropics and are strongly influenced by their hydrology (Spalding and Leal, 2021). This includes influxes of freshwater and saltwater controlling salinity, as well as associated inputs of nutrients, organic matter, trace elements and the biogeochemical regime (Lacerda et al., 2022). Their occurrence in sheltered or protected coastlines is due to the impact of wave energy, which can result in erosion of sediments and mangrove degradation and loss. However, with the Intergovernmental Panel on Climate Change (IPCC) predictions suggesting that the frequency and intensity of tropical storms is set to increase, this may alter the dynamics between stable mangrove coastlines and unvegetated higher energy coastal systems, particularly in areas where tropical storms have not previously been a common occurrence (Ward et al., 2016). Recent hydrological alterations have been recorded to occur throughout the distribution of mangroves including sea level rise, which can be exacerbated by local factors including land subsidence, sediment starvation due to upstream or offshore removal/trapping/diversion of sediment flows (Balke and Friess, 2016), or sub-surface sediment collapses linked to soil degradation, root death or bioturbation (Arnaud et al., 2023).

Other factors that can influence mangrove hydrology can be diversion or damming of watercourses, alterations through land use and land cover changes, and changes in inputs of freshwater (either through increased precipitation or drought). In semi-arid or arid mangrove areas, drought can result in tree die off in already stressed mangrove ecosystems, such as has been recorded in Iran and NE Brazil (Mafi-Gholami et al., 2018, Nogueira Servino, 2018) (Figure 1). In areas where there are substantial freshwater inputs and ecotones between tidal freshwater forests (TFF) and mangroves occur, drought can result in mangroves encroaching into former TFF ecosystems, either altering plant community composition or entirely replacing it as has been recorded in the freshwater mangroves of the Amazon delta (Guimarães et al., 2010, Ward et al. in review).

Mangroves are constrained to the tropics as a result of their thermal limitations and will typically only occur where mean monthly temperatures of the coldest month are ~20°C, and freeze temperatures do not occur, although this is regionally variable (Quisthoudt et al., 2012). As a result, temperature increases globally are likely to influence the distribution of mangroves, particularly at the extremes of their latitudinal distribution. Recent studies have suggested that while there is likely to be an increase in expansion away from the tropics, this is not being seen yet, although there is an increase in tree density at some of these latitudinal limits (Saintilan et al., 2014). While this may seem as a positive, it should be noted that this is typically at the expense of already established saltmarshes with their own suite of ecosystem services (Osland et al., 2022). Furthermore, while temperatures are set to increase globally, in some areas, mangroves are already at their peak thermal efficiency from a biological perspective and any increases in temperature, particularly for prolonged periods would likely result in decreases in photosynthetic activity and/or plant death, with resultant impacts on plant productivity and ecosystem service provision (Figure 1) (Friess et al., 2022).

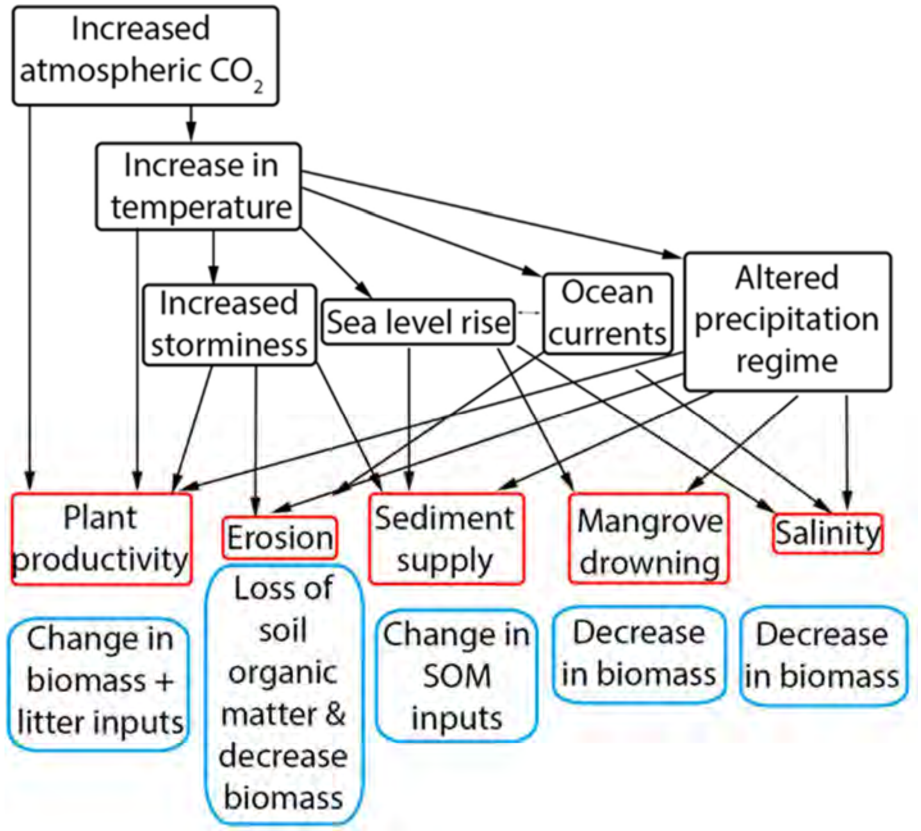

Figure 1. Impacts of climate change on mangrove ecosystems and their impacts on carbon stocks.

2. Factors influencing mangrove blue carbon potential

Mangrove ecosystems have been widely recognized for their role in sequestering CO_2 from the atmosphere and storing it in their above ground and below ground biomass and soils, playing a pivotal role in mitigating climate change, commonly referred to as blue carbon (Chmura et al., 2003, Pendleton et al., 2012). They also act as long-term sinks for carbon from allochthonous sources, both terrestrial and marine (Veettil et al., 2024). A range of factors influences the efficiency and capacity of mangroves to act as blue carbon sinks and these can be categorized into anthropogenic, ecological and abiotic influences.

2.1. Anthropogenic factors

Anthropogenic influences can have a profound impact on mangrove ecosystem functioning, affecting their ecological health and carbon storage and sequestration potential (Lacerda et al., 2022; Ferreira et al., 2024). Deforestation and subsequent conversion to agriculture, aquaculture and urban development has historically been one of the most significant threats to mangroves (Lacerda et al., 2022; Ximenes et al., 2024). Studies indicate that approximately 35% of global mangrove area has been lost since the 1980s, primarily due to conversion for shrimp farming and urban expansion (Lacerda et al., 2021). This loss not only reduces the area available for carbon sequestration but also releases stored carbon, resulting in these ecosystems becoming greenhouse gas emitters (Lacerda et al., 2021, Ward et al., 2022).

Moreover, pollution from agricultural runoff, industrial discharges and plastic waste can degrade mangrove health, leading to reduced photosynthetic activity with resultant impacts on primary productivity and biomass accumulation (Celis Hernandez et al., 2021). The introduction of invasive species can also disrupt local ecosystems, leading to changes in species composition and, consequently, carbon storage dynamics (Guimarães Sampaio et al., 2021). Climate change further exacerbates these problems by increasing the frequency of extreme weather events, such as storms and flooding, which can physically damage mangrove forests and alter their delicate ecosystems (Mafi-Gholami et al., 2018; Celis Hernandez et al., 2021).

Effective management and conservation strategies, including restoration efforts and sustainable land-use practices, are crucial for maintaining broader mangrove ecosystem service provision including as blue carbon store (Veettil et al., 2018). The restoration of degraded mangrove areas has been shown to enhance carbon storage. For example, a mangrove restoration potential in Southeast Asia could be in the order of up to 4,081,325 ha, resulting in an estimated sequestration potential of 1839 kg $CO_2.ha^{-1}.y^{-1}$ (Syahid et al., 2024). Successful restoration involves selecting appropriate species, ensuring proper hydrology and engaging local communities in the process, in part due to their knowledge of the local systems and to ensure ongoing support and success following restoration campaigns (Veettil et al., 2018).

Integrating sustainable land-use practices is crucial for maintaining the health of mangrove ecosystems, whether it be derived from environmental education campaigns or through policy-led protections (Akram et al., 2023). The use of buffer zones, where development is restricted, can help protect mangroves from urban encroachment and agricultural expansion. For instance, the establishment of protected areas around the Sundarbans mangrove forest in Bangladesh was shown to significantly reduce deforestation rates and enhance carbon storage (Rahman et al., 2017). Recent changes to the Forest Code in Brazil (the main legal protection covering mangroves in the country) have decreased protections to adjacent ‘buffer’ zone ecosystems such as *apicuns* (salt flats) and opened them up to development (Lacerda et al., 2022b).

Engaging local communities in the management of mangrove resources can lead to more sustainable practices. Programs that educate communities about the benefits of mangroves, such as carbon storage, fisheries support and coastal protection, can foster stewardship (Veettil et al., 2018). A coastal community restoration project in Indonesia has successfully involved local fishermen in mangrove rehabilitation, leading to increased fish populations and improved livelihoods (Debrot et al., 2022).

Effective policy frameworks are necessary to support mangrove conservation efforts, whether driven by nature-based solutions or natural capital renewal programmes, or biodiversity/carbon crediting agendas. Governments should enact policies that incentivize or enable the protection and restoration of mangroves, including carbon credit systems that reward landowners or land stewards for maintaining these ecosystems, particularly relevant following the agreement announced at the recent Conference of the Parties 29 (COP29) in Baku to push towards the development an international carbon crediting system. However, it is important to highlight that carbon crediting systems should be developed to take into account a range of factors and promote synergistic development of other ecosystem services and biodiversity; also, they should consider traditional land stewards (e.g. indigenous people), and should not be considered a primary driver to address climate change.

Maintaining carbon stores in mangroves requires a holistic approach that combines restoration efforts, sustainable land-use practices, community engagement and supportive policies. By implementing these strategies, we can ensure the continued health and functionality of mangrove ecosystems, enhancing their role in climate change mitigation.

2.2. Ecological factors

The ecological characteristics of mangrove forests play a crucial role in their ability to sequester carbon. The diversity of mangrove species contributes to the overall productivity of the ecosystem. Different species exhibit varying growth rates, biomass accumulation and root morphologies, all of which are important for carbon storage (Bai et al., 2021). For instance, *Rhizophora* species tend to have extensive root systems that stabilize sediments and enhance carbon accumulation compared to other species with less developed root structures (e.g. *Conocarpus erectus* or *Laguncularia racemosa*).

Additionally, the structural complexity of mangrove forests influences their carbon dynamics. Dense mangrove canopies and above ground root networks provide a three-dimensional structure and thus a range of habitats for a range of epifauna and epiflora, which can contribute to nutrient cycling, enhancing productivity and carbon storage (Ferreira et al., 2024). Furthermore, the connectivity between mangroves and adjacent ecosystems, such as tidal freshwater forests, seagrasses and coral reefs, facilitates nutrient exchange and supports overall ecosystem health, which is essential for maintaining carbon sequestration (Saavedra-Hortua et al., 2023; Ward et al., in review).

Soil microbiota, including bacteria and fungi, play a crucial role in the carbon cycling processes within mangroves and their activities influence carbon storage and sequestration, impacting the overall health and resilience of these coastal ecosystems.

Microbial communities in mangrove sediments are essential for the decomposition of organic matter facilitating carbon cycling. Sulphate-reducing bacteria, enhance the breakdown of complex organic compounds, thereby releasing nutrients and promoting plant growth, which contributes to increased biomass carbon storage (Wang et al. 2023). Fungal communities, such as mycorrhizal fungi, form symbiotic relationships with mangrove roots, providing and enhancing access to nutrients in nutrient-limited mangroves, and thus promoting carbon assimilation (D'Souza, 2016). Certain microbial groups are involved in methanogenesis, which can influence carbon fluxes in mangrove ecosystems (Lacerda et al., 2022a). While methane is a potent greenhouse gas, understanding its production by methanogenic biota in mangrove sediments is vital and environmental factors, such as salinity and organic matter availability, can significantly affect methane emissions within mangrove ecosystems (Das et al., 2021). A diverse community of soil microbiota contributes to mangrove ecosystem resilience by enhancing nutrient cycling and plant growth. Increased microbial diversity has been linked to improved carbon storage capacity, as diverse microbial communities can utilize a broader range of organic substrates (Wang et al., 2023). This diversity is crucial for maintaining ecosystem stability in the face of environmental changes such as climate change. Soil microbiota play an essential role in the carbon storage and sequestration processes in mangrove ecosystems through their impact on organic matter decomposition, soil stabilization, methane production and overall ecosystem resilience, although microbial community dynamics are often poorly understood.

2.3. Abiotic factors influencing mangrove blue carbon

The capacity of mangroves to sequester carbon is significantly influenced by a range of abiotic factors including edaphic characteristics (e.g. soil pH and granulometry), hydrology (frequency and duration of inundation), salinity, temperature and nutrient availability (Ward et al., 2022). Understanding these factors is essential for effective management and conservation strategies aimed at enhancing or maintaining carbon storage and rates of sequestration.

2.3.1. Soil characteristics

Soil properties are pivotal in determining the carbon storage capacity of mangrove ecosystems. Mangrove soils are typically waterlogged and anaerobic, which slows down the decomposition of organic matter (Veettil et al. 2024). The predominantly anaerobic soil environment supports the accumulation of carbon-rich peat (Chmura et al., 2003). The texture and composition of the soil also play significant roles; clay-rich soils typically have higher carbon storage potential compared to sandy soils due to greater retention of organic matter (Ward et al., in review). Additionally, soil depth is strongly related to total carbon stocks, as deeper soils typically store more carbon (in some cases over millennial time scales) (Chmura et al., 2003; Pendleton et al., 2012). However, usually blue carbon studies only consider the upper layer (0.3−1.0 m) because this is the layer that is most likely to result in losses or emissions of carbon when degraded or impacted by stressors and sedimentation rates factors influencing them (Maxwell et al., 2024). Sedimentation plays a dual role; while adequate sediment supply can enhance carbon storage (particularly through inputs of particulate organic matter but also through physical methods of protection from breakdown and loss of carbon to the system), excessive sedimentation can smother mangrove roots and inhibit growth with associated knock-on effects on productivity and carbon assimilation to biomass (Nardin et al., 2021).

2.3.2. Hydrology

The hydrological regime, including tidal inundation patterns, freshwater input, and in some cases monsoonal inundation patterns (e.g. the largest mangrove ecosystems in the world are Amazon Macrotidal Mangrove Coast in Brazil and the Sundarbans in India/Bangladesh) influences both the growth of mangroves and their carbon sequestration capacity. Tidal flooding is essential for nutrient delivery and sediment deposition, which are crucial for mangrove health (Lacerda et al., 2021; Ward et al., 2022). However, altered hydrology due to human activities like dam construction, land reclamation and water diversion can disrupt natural tidal patterns, leading to stress on mangrove populations and reduced carbon storage (Balke and Friess, 2016). Regular tidal flows help flush out saline water and provide nutrients, thus supporting mangrove growth.

Hydrological influences on carbon sequestration and storage can be linked to a range of factors such as:

1. Water availability: The saturation of soil with water directly affects plant productivity and as such growth rates of mangrove trees and their ability to sequester carbon. Soil hydrology regulates root respiration and decomposition rates, influencing the overall carbon storage capacity of mangrove ecosystems through biomass allocation (Castaneda Moya et al., 2013).

2. Salinity: Mangroves are adapted to varying salinity levels, which can influence species composition and productivity. Changes in salinity due to hydrological alterations, such as freshwater input from rivers, can significantly impact carbon dynamics. Lower salinity levels typically enhance growth rates and carbon sequestration (Ray et al., 2011), whereas higher salinity levels such as those found in arid or semi-arid mangroves result in stressed conditions leading to lower plant productivity and decreased biomass (Lacerda et al., 2024).

3. Nutrient transport: Hydrology also governs the transport of nutrients essential for mangrove growth. Nutrient enrichment from freshwater inflows can enhance primary productivity, resulting in increased carbon uptake (Ward et al., in review). Conversely, excessive nutrient loading can lead to detrimental effects, such as harmful algal blooms, which can disrupt ecosystem function (Lovelock et al., 2009).

4. Carbon deposition: Sediment accumulation in mangrove ecosystems is a critical component of carbon storage and the bulk of this is from allochthonous sources (Ward et al., 2022). Hydrological processes, including tidal fluctuations and sediment transport, determine the rate of allochthonous sediment and organic material deposition (Veettil et al., 2024). The interplay between hydrology and sediment dynamics is crucial for maintaining the long-term carbon storage potential of mangrove forests.

2.3.3. Salinity

Salinity levels are a fundamental abiotic factor affecting mangrove ecosystems. Mangroves are uniquely adapted to tolerate saline conditions, but excessive salinity has been found to inhibit growth and reduce productivity (Ward and Lacerda, 2020). Different species exhibit varying tolerance to salinity, which can influence species composition and forest structure at a variety of scales within an estuary and their response to larger scale climatic drivers (Guimarães et al., 2010). High salinity can lead to physiological stress in mangroves, resulting in decreased biomass and, consequently, lower carbon sequestration rates in the biotic component of the ecosystem (Figure 1) (Lacerda et al., 2022a).

2.3.4. Temperature

Temperature is an important factor governing carbon dynamics within mangrove ecosystems, affecting processes such as photosynthesis, respiration and decomposition rates (Ward et al., 2016). Understanding the effects of temperature on these processes is essential for assessing the role of mangroves in global carbon cycles and their response to climate change (Figure 1) (Friess et al., 2022).

Mangrove ecosystems, located in tropical and subtropical coastal regions, are characterized by high productivity and significant carbon storage potential, and temperature provides a limitation to their distribution (Saintilan et al., 2014). Understanding how temperature variations impact carbon dynamics is crucial for predicting the future mitigation potential of these ecosystems in the face of climate change.

Photosynthetic rates in mangroves are highly temperature-dependent. Optimal temperatures (typically around 30°C) enhance photosynthetic efficiency, leading to increased carbon fixation (Akaji et al., 2019). However, extreme temperatures can inhibit photosynthesis and result in plant death (> 40°C), reducing carbon uptake and potentially leading to large scale die-off events in mangrove species (Figure 1) (Ward et al., 2016) particularly, where impacted by other stressors. Temperature also affects respiration rates in mangrove plants and soils. Increased temperatures generally lead to higher respiration rates, resulting in greater carbon loss as CO_2. This process can significantly offset gains from photosynthesis, particularly during extended periods of high temperatures (Chmura et al., 2003).

The decomposition of organic matter in mangrove sediments is influenced by temperature, which affects microbial activity. Warmer temperatures typically increase microbial metabolic rates, accelerating the decomposition of labile organic material stored in sediments (Alongi, 1988).

Rising global temperatures due to climate change are likely to alter mangrove carbon dynamics. Alterations in temperature can lead to shifts in species composition, affect growth rates, and alter the balance between carbon uptake and loss (Friess et al., 2022). Understanding these dynamics will be important for blue carbon accounting, as this will be linked to the rate and extent of changes and potentially result in some carbon stores being reduced or becoming emitters.

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Chapter 4

Mangroves mitigate pollutant transfer through estuaries

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1. Introduction

1.1. The critical role of coastal areas to pollutant transport

Major human civilizations have developed along waterways and coastal zones, considered ideal locations for dumping their by-products. Although evidence of harmful effects on the environment has existed since the period of the great Mediterranean civilizations, it was only with the industrial revolution and the consequent urbanization of the population that such effects became alarming. The growth of the human population, intensification of urbanization and industrialization that occurred after the industrial revolution, followed by an accelerating increase in demand for primary and secondary goods, led to an unprecedent use of natural resources. This overexploitation process inevitably resulted in a growing degradation of natural ecosystems in such a large scale that from the 1960s onwards, the growing pressure from public opinion led to the enactment of national laws and international conventions, as well as public policies supporting the investment in 'clean' technologies. These actions together started changing the scenario in the last 50 years, with the result being a drastic reduction in contaminant emissions and in the recovery and management of contaminated areas. Unfortunately, though, these initiatives were much slower in developing countries, usually located in tropical regions. In addition, despite the relative success of the measures adopted, the reduction of nutrients and of non-degradable contaminant emissions into the environment has not decreased in the same proportion as the introduction of control technologies and regulatory policies. Therefore, these pollutants still trigger eutrophication and contamination in many world regions, particularly in coastal areas.

1.2. A summary of mangrove response to pollution

Irrespective of being local or distal, anthropogenic activities release nutrients and pollutants and/or have the potential to remobilize them (See Chapter 1, for a summary). Estuaries, where mangroves thrive, are typical accumulation environments of sediments, nutrients and pollutants brought in from watersheds (Botello et al., 2017). Therefore, it is pivotal to understand how mangroves face this excess of potentially toxic substances and eventually mitigate their effects, making this ecosystem service of significant environmental and economic importance globally.

The considerable increase in anthropogenic mobilization of metals and nutrients over the last 200 years has resulted in the creation of large 'stocks' accumulated in natural 'sinks', such as soils and aquatic sediments. Also, marine minimum oxygen zones (MOZ) formed worldwide lead to a constant concern about possible effects on the biosphere, including humans, due to increased chronic exposure to metals and the deterioration by eutrophication of areas relevant to coastal fisheries and biodiversity conservation. Several natural mechanisms are capable of remobilizing, transporting and accumulating metals and nutrients to potentially harmful levels, even from the currently more controlled emissions. Recent studies strongly suggest that these mechanisms are intensifying due to global warming (Lacerda et al., 2020; Hazri et al., 2024).

Practically all confined and semi-confined coastal areas in the world have some degree of contamination and eutrophication. Some peri-urban bays and coves adjacent to large population centres are totally degraded, requiring complex, expensive and careful environmental interventions. The ecological characteristics of other coastal areas, may still be relatively little modified but given the environmental pressures occurring in their drainage basins, they are permanently subject to environmental hazards. In these areas, it is essential to permanently monitor their environmental conditions and strictly control potentially polluting activities in their drainage basins. Thus, hardly any ecosystem is currently still in its pristine state (Rockström et al., 2022), and several authors suggest that both environmental legislation and programs for the conservation, management and recovery of these ecosystems should address this reality.

Natural ecosystems are already subject to different degrees of environmental pressure and consequently, have decreasing capacity to withstand extra pressures arising from the remobilization of contaminants accumulated over decades in the coastal region (Lacerda, 2003). In the tropics, almost all coastal regions under this scenario are colonized extensively by mangroves. Figure 1 (modified from Salomons and Stigliani (1995) and Lacerda (2003), and adapted to mangrove ecosystems), shows a conceptual analysis of the impacts of continuous pollutant loads on their mobilization in mangrove ecosystems. Like all other coastal ecosystems, we can find mangroves with different support capacities and resilience to environmental pressures. In regions only mildly impacted by environmental pressure, mangroves are still very far from exhausting their carrying capacity, that is, they are still capable of acting as a sink for pollutants, reducing and slowing their transport and mobilization (Figure 1; Q_1, C_1). However, when approaching their maximum support capacity (consequence of an increase in environmental pressures), the pollutant retention capacity of mangroves decreases exponentially and small variations in the input of pollutants result in large mobilization and transport (Figure 1; Q_2, C_2), increasing the exposure of biota and human populations who depends on fisheries and other natural products. When partially or totally degraded, mangroves may even lose their accumulating capacity completely, turning into exporters of excess nutrient and metal pollutants and causing significant impacts on adjacent environments and their biological communities.

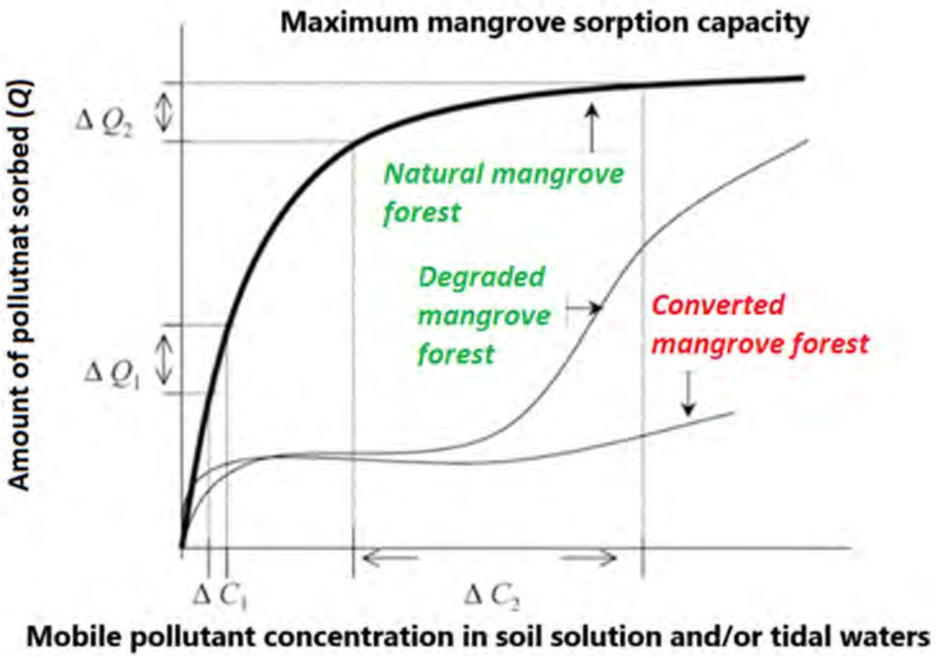


Figure 1. Theoretical response of mangroves to environmental pressures. Similar loads of a given pollutant (y-axis) result in increasing mobilization of it in mangrove pores and tidal and creek waters (x-axis). Adapted from Salomons and Stigliani (1995) and Lacerda et al. (2003).

1.3. Nutrient filtering capacity

Mangroves play a key role in nutrient cycling in tropical estuaries, mostly as a sink, but sometimes as a source of N and P to adjacent environments. As such, they can play a significant role in mitigating the impacts of excess emissions and/or remobilization due to local anthropogenic drivers and global environmental changes (Valiela et al. 2018). The root morphology, high tree densities and elevated primary production of healthy mangroves, efficiently trap suspended particles coming in with the tides, resulting in high sedimentation rates and high litter deposition, eventually burying organic matter into anoxic sediments and thus decreasing organic matter degradation and nutrient release. However, land-use practices upstream of mangrove areas may represent a chronicle threat due to increasing inputs of total N and P to mangroves following the increasing intensity of anthropogenic drivers (Sanders et al., 2014). This may negatively affect mangrove forests health and their capacity to filter materials from tidal waters; it may even reverse mangrove's accumulating role of mangroves, turning them into sources of nutrients to coastal waters.

Reduction in canopy health was observed, for example, in mangroves downstream of shrimp farms. The normalized difference vegetation index (NDVI), which compares photosynthetic activity to canopy structure of forests, strongly associates mangrove degradation with increasing shrimp farm area (Alatorre et al., 2016). In the Jaguaribe Estuary, located in the semi-arid equatorial coast of Brazil, Marins et al. (2020) showed a 15% reduction in the NDVI between 2003 (0.78) and 2008 (0.65) followed by a faster 70% reduction in 2017 (0.20) coinciding with an increase in shrimp pond area from 340 ha (2003) to 1600 ha (2008) and to ~3400 ha (2017). This increased the annual load of P to approximately 69 tons (Marins et al., 2007, 2011; Lacerda et al., 2021). The ability of mangroves receiving these effluents, to accumulate P was reduced by over 50% (Figure 2) relative to pristine mangroves in estuaries not affected by shrimp farm effluents, which accumulated over 93% of the total load of suspended sediments, total P, soluble reactive P and particulate P, entering the system by tides. On the other hand, the mangrove area affected by shrimp farms accumulated only 54% of the suspended sediments and particulate P loads and was particularly inefficient in retaining total P (46%) and soluble reactive P (38%). This 'nutrient leak' triggered algal blooms and eutrophication in the estuarine waters adjacent to the shrimp farm impacted estuary (Marins et al., 2020).

Phosphorus fluxes in mangroves are usually small (Holmer et al., 2001), even limiting mangrove growth, thus the role of mangroves as a net source or sink for this nutrient may depend on the magnitude of the total P input, as suggested by the results from NE Brazil. There, anthropogenic sources, mostly aquaculture, contribute to excess P entering the mangrove system at the shrimp farm-impacted estuary, whereas in the pristine estuary P inputs are mostly from natural sources, and probably limit tree growth (Marins et al., 2020). As such, the P balance in the pristine site agrees with a P-limitation response of mangroves as proposed by Feller et al. (2007), implying higher P retention rates under low P availability. This also supports the rapid increase in mangrove cover and biomass observed by Lacerda et al. (2007), in this pristine estuary. However, at the impacted Jaguaribe site, mangrove area and biomass remained relatively constant or even decreased in the past 20 years (Godoy et al., 2018). Although several studies suggest a role of mangroves as P sinks (Tappin, 2002), the results from NE Brazil estuaries suggest that P import rates are higher than many previously reported values (0.01 to 0.03 $kg.ha^{-1}.day^{-1}$) that have been observed in humid areas with high P inputs from rivers (Wörsten, 2003), but similar to those observed in other semi-arid sites (Sanchez-Carrillo et al., 2009). This role as a sink for P, as observed in the pristine site, but not in the aquaculture-affected mangrove, suggests that mangroves are useful sites for the disposal of nutrient-rich effluents from human activities, including aquaculture, currently the most significant driver of environmental impacts on mangroves (Lacerda et al., 2019, 2021). However, the P balance observed in the impacted Jaguaribe mangrove and in other impacted sites suggests that eutrophication reduces the resilience of mangroves to environmental stress, eventually increasing tree mortality (Lovelock et al. 2009) by augmenting foliar N contents and thus attractiveness to insect herbivory (Lacerda et al., 1986), and reducing foliar base cations as well as the availability of P in soils (Fauzi et al. 2014).

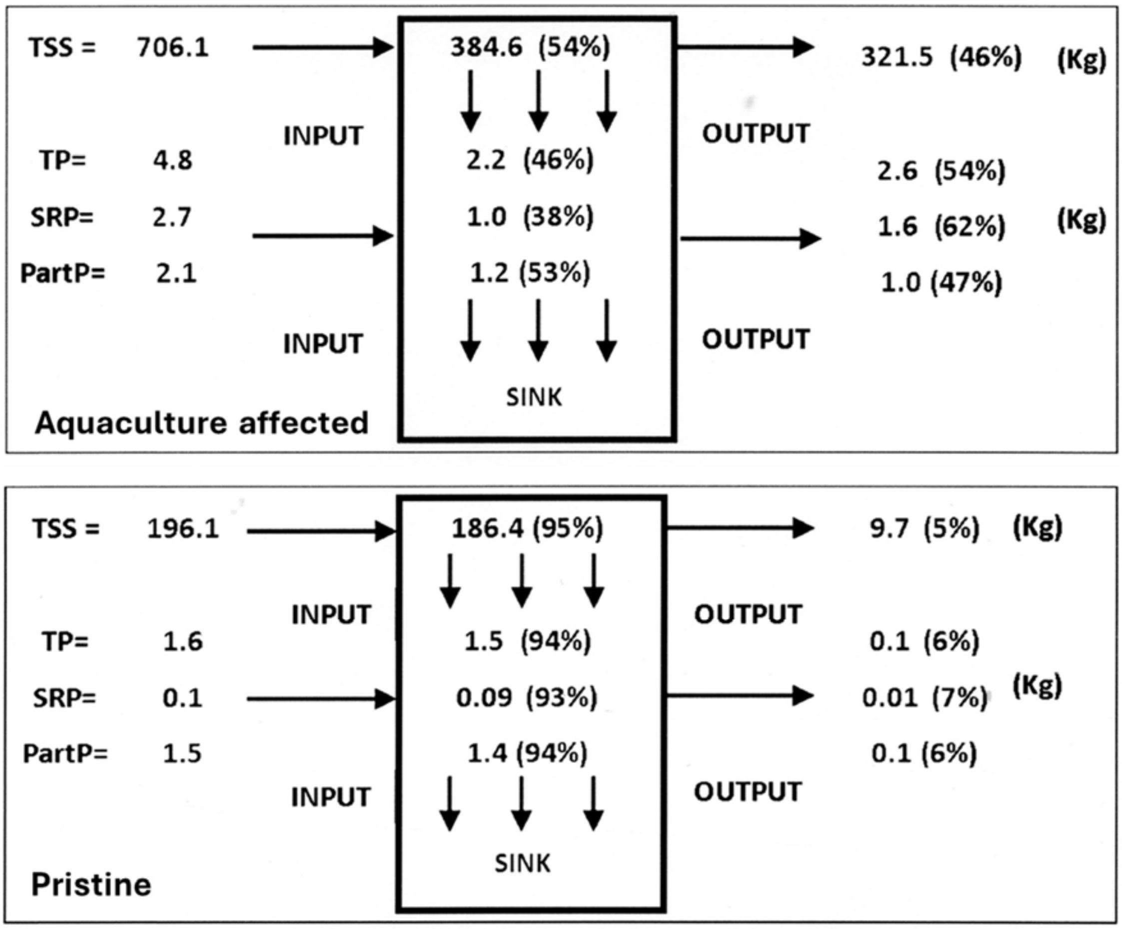


Figure 2. Diagram comparing the instantaneous fluxes of total suspended solids (TSS) and phosphorus fractions: SRP = soluble reactive phosphorus, PartP = particulate phosphorus and TP = total phosphorus, to a mangrove receiving shrimp farming effluents, at the Jaguaribe estuary, and a pristine one at the Pacoti estuary, both in NE Brazil. Fluxes are in kg.h^{-1}.

Excess nutrients, traditionally from anthropogenic sources such as poor sanitation, sewage treatment and inadequate waste disposal, trigger eutrophication in many mangrove-dominated estuaries. More recently, however, rapidly expanding, intensive shrimp farming has increased enormously the eutrophication pressure on mangrove ecosystems due to excess nutrients (Lacerda et al., 2019). Therefore, the idea of using mangroves as filters to human-derived nutrients is quite appealing. The mangrove's role as a nutrient filter is based on actual measurements of net imports of nutrients by mangroves, which show that only a fraction of the nutrient input entering the forest is exported back to adjacent coastal areas (Sanchez-Carrillo et al., 2009, Silva et al., 1998). The same picture was obtained by modelling nutrient balances based on actual measurements of nutrient concentrations in waters and their changes as a function of dilution (Bin and Dushof, 2004). Therefore, mangroves seem to actively immobilize nutrients, either accumulating them in mangrove biomass and/or in sediments.

In a seminal paper on the use of mangroves as filters of shrimp farm effluents, Robertson and Phillips (1995) estimated that 2.8 to 21.7 ha of mangroves could filter the P load from 1.0 ha of shrimp pond. Unfortunately, the results summarized from the NE Brazil sites discussed above strongly suggests that this role as a filter is occurring only in pristine mangroves. When considering the immense eutrophication potential of existing and expanding areas of shrimp ponds in NE Brazil (also worldwide), the filtering capacity proposed by Robertson and Phillips (1995) is already surpassed by a factor of 6 to 48 in the impacted site, although the P accumulation capacity is still underutilized in the pristine site by a factor of 0.1 to 0.6. This agrees with the proposed scenario presented in Figure 2 and strongly suggests that mangrove P accumulation capacity is significantly decreased with increasing P inputs, thus limiting the mangrove's potential as a natural barrier of nutrient transport through the continent-ocean interface. Therefore, as in other marine ecosystems, mangroves may also exhibit a threshold response. Although this threshold is very difficult to predict in face of increasing cumulative stress, including shrimp farming effluents (Crain et al. 2008), this needs to be taken into consideration when managing mangrove areas already under stress by anthropogenic activities, in order to mitigate the impacts of excess nutrient loads. Shrimp pond effluents are enriched not only in nutrients but also in trace elements such as copper (Cu) and mercury (Hg) that are present in aquafeed and chemicals used in shrimp production. Emission factors of N, P, Cu and Hg are higher than all other anthropogenic sources in most tropical estuarine regions and concentrations in excess of natural levels are ubiquitous in adjacent tidal creek waters (Lacerda et al., 2021). The presence of toxic metals in shrimp farm effluents may eventually contribute to decrease mangrove health and to diminish the efficient of trapping nutrients.

1.4. Mitigating heavy metal pollution

Mangrove soils are characterized by fine sediment particles, with a preponderance of the silt-clay fraction (< 0.0062 mm). The flooding of these soils is dominated by the ebb and flow of the tides that bring suspended materials as well as environmental contaminants. When flooding occurs, tidal water interacts with the tangle of aerial mangrove roots decreasing the sediment transport capacity and accumulating sediments within the forest. Most mangrove stands have high sediment accretion rates (SARs), in the order of 0.07 to 0.61 cm.year^{-1}, but these rates can reach extreme values of over 0.7 cm.year^{-1} under specific situations such as in urban mangroves or in areas of enhanced sediment transport (Table 1). As a result of high SARs, the deposited particles and their contaminant load are quickly buried in mangrove sediments.

Table 1. Sediment accretion rates (SAR) derived from ^{210}Pb dating (cm.y^{-1}) measured in mangroves from different parts of the Brazilian coast.

Region	Sites	Mean SAR (cm.y^{-1})	Reference
Brazilian semi-arid coast	Pristine	0.15–0.31	Ward et al. (2023)
	Urban	0.71	
Amazon humid macrotidal coast	Pristine	0.07–0.38	
	Urban	0.71	
Southeastern Brazil humid coast	Pristine	0.1–0.2	Smoak and Pachineelam (1990)
Central northeastern Brazil humid coast	Pristine	0.61	Hatje et al. (2021)
	Extreme	3.1	
Southern Brazilian humid coast	Pristine	0.25–0.39	Sanders et al. (2010)

The typically high primary productivity of mangroves yields the highest mean litterfall rate in high-SAR riverine and deltaic forests and results in a large accumulation of organic matter in soils (11.5 Mg C.ha^{-1}.year^{-1}; Ribeiro et al., 2019) *via* litterfall, leading to high soil organic matter content (in the order of 5 to 15%). Bacterial decomposition of organic matter and the fine granulometry of sediments result in a rapid consumption of the oxygen present in these soils, making them anoxic. Therefore, deposited particles are rapidly transferred to an anoxic environment. This mechanism controls much of the dynamics of metal pollutants in mangroves, accumulating them in the sediment and making mangroves into true natural ‘filters’ that hinder the migration of metals to the adjacent coastal environments. The processes involved the mechanisms of metal transport to mangroves and the biogeochemical environment they find in the mangrove soils after sedimentation.

Metals reach mangroves through atmospheric deposition, river and tidal transport. In regions influenced by adjacent industries, metals can be released directly into mangrove forests from erosion and/or leaching of tailings. Metals can also be brought directly to mangroves by rivers or be redistributed throughout the coastal region and transported to the mangroves by tidal currents or by the flooding of mangrove areas when the flow of rivers and canals is blocked by the tide. They are mainly adsorbed onto suspended material, and Fe and Mn oxyhydroxides (Lacerda et al., 1988, 2022). Once deposited within the forest, the accumulation of suspended particulate matter and its associated metals will be influenced by the physical and chemical conditions of the interstitial water and sediments (such as particle size, organic matter content, pH and redox conditions), as well as by the presence of different vegetation cover (Chiu and Chou, 1991; Tam et al., 1995; Lacerda et al., 2022).

A high SAR, in conjunction with the accumulation of organic matter and high flooding frequency result in the rapid consumption of oxygen in the interstitial waters of the sediments. This supports two different mechanisms that immobilize metals in mangrove soils (Figure 3). A first subsystem is dissimilatory sulphate reduction whereby, all available oxygen is used up by aerobic microorganisms in the sediment, which then become quiescent or die. Under such condition there is a proliferation of anaerobic microorganisms, that respire organic matter along with products of its previous degradation as a substrate, using oxidized components of sediments as electron receptors in respiration; examples are, the reduction of ferric ions (Fe^{3+}), nitrates and Mn^{4+}. These electron receptors are quickly exhausted, and the remaining most abundant supply of electron receptors is sulphate, which is present in large quantities in sea water. Thus, in mangroves, sulphate reduction is the main mechanism of organic matter respiration by anaerobic microbiota in the sediments. The by-product of this metabolism is a significant increase in sulphide concentrations in interstitial waters, which directly influences the solubility of chemical elements and allows the precipitation of pyrites and the co-precipitation and accumulation of chalcophile elements (including toxic metals of environmental significance such as Cd, Pb, Hg, Zn and Cu), and strongly reducing their bioavailability to mangrove plants and associated biota, and acting as a significant sink of these pollutants (Ferreira et al., 2007; Deborde et al., 2015). Framboidal pyrite crystals measuring ~40 microns abound in mangrove soils (see scanning electron microscopy photograph; in Figure 3). An X-ray analysis of this framboid reveals the dominant presence of sulphur and iron, elements forming the most common type of pyrites. Through the same technique it is also possible to reveal the presence of chalcophile toxic heavy metals, such as Cu, in the framboid (Aragon and Miguens, 2001; Ferreira et al., 2021; Lacerda et al., 2020, 2022). Maintaining the reducing conditions in sediments keeps pyrites crystalized and hence, a permanent sink for metal pollutants.

A second subsystem that results in metal immobilization (also resulting from anaerobic conditions), is the accumulation of sulphides in sediment porewater that may exceed the tolerance threshold of mangrove trees. This induces these plant species to release oxygen through the roots resulting in Radial Oxygen Loss (ROL), thus oxidizing potentially toxic reduced compounds in the rhizosphere. In addition, ROL precipitates ferric ions (Fe^{3+}) forming iron plaques, which co-precipitates metals as iron oxyhydroxides, in the rhizosphere (Figure 4A), as is evident by the higher percentage of total Cu sediments that is associated with iron plaques (Figure 4B). This mechanism contributes to a further reduction in metals availability for plant uptake and translocation from roots to other plant organs (Chen et al., 2020; Lacerda et al., 2024).

Similar to eutrophication, pollution by metals is ubiquitous in most tropical coastal areas, therefore the capacity of mangroves to accumulate metals under non-bioavailable chemical forms, i.e., as precipitated sulphides or adsorbed onto iron plaques at the outer surface of roots, as has been suggested as a process for sequestering and immobilizing metal pollutants and hampering their transport through estuaries. The unique characteristics of mangrove plants that avoid, tolerate, and/or adapt to heavy metal toxicity highlight the potential of mangrove plants for phytoremediation, phytoextraction and bioaccumulation of metals in coastal areas (Rahman et al., 2024). This 'bioengineering' control approach to mitigate metal pollution has been attempted with relative success.

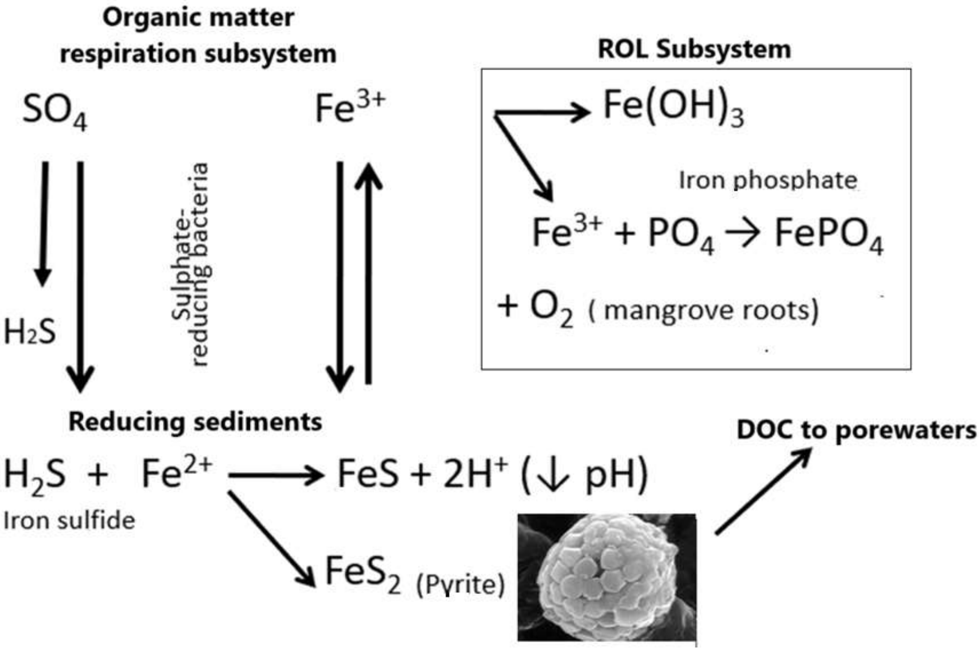
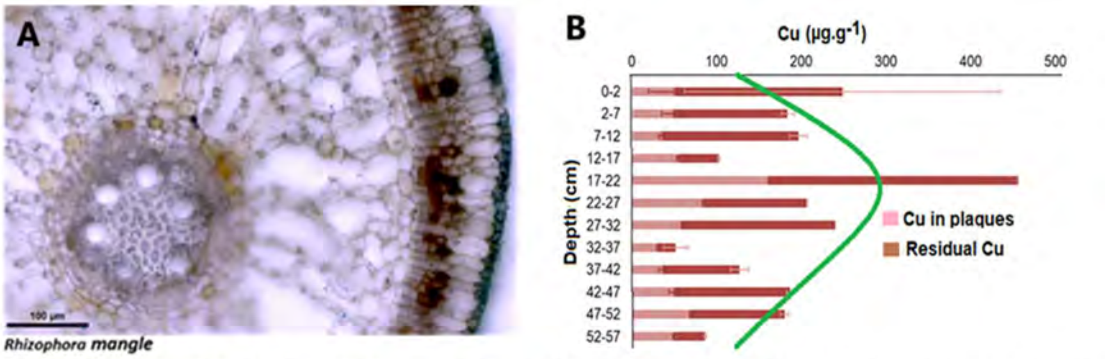


Figure 3. The sulphate reduction metabolism of mangrove sediments is used by bacteria to extract oxygen from sea water sulphates to oxidize organic matter. This relatively inefficient metabolism produces dissolved organic complexes, that may form organo-complexes with metals and export them to adjacent waters, and sulphides, that precipitate chalcophile metals in sediments permanently if conditions remain anoxic. Radial Oxygen Loss (ROL) precipitate iron and metals onto the surface of roots, immobilizing them as iron oxides.

The vertical distribution of selected heavy metals in sediments from forested and bare areas within degraded mangrove stands surrounding a landfill in Guanabara Bay, southeastern Brazil is shown in Figure 4C. Guanabara Bay is one of the most polluted bays in the tropics; the landfill receives ~20,000 tons of solid wastes per day. Urban solid wastes are a significant source of metals to the environment (Nriagu, 2019). The capacity of the mangrove rhizosphere to immobilize metals is clearly shown by the sharp increase in total metal content in the sediment layers with the highest root biomass, as is seen in the forested site (Figure 4, green box). Alternatively, metal profiles in the sediments from the bare, degraded mangrove stands, show a steadily increase in concentrations towards the surface, suggesting transfer to pore and surface waters (Lacerda et al., 2000). Therefore, the presence of mangroves significantly decreases metal transport to Guanabara Bay waters thus protecting coastal biological resources from metal contamination.

In many countries replanting or rehabilitating mangroves surrounding contaminated areas (phytoremediation) has proven to be a strategy to immobilize metals in the environment without disturbing the ecosystem serving as an efficient tool to mitigate pollution (Rahman et al., 2024). For example, chromium (Cr) retention by mangroves in the Thi Vai catchment, south Vietnam was shown to be very effective under pristine conditions, but modelling the impact of tree mortality in this Vietnamese estuary suggests there is a rapid loss of the capacity to sequester metals following mangrove degradation (Nguyen et al., 2020). This result is similarly to what was observed in the P retention capacity of mangroves in NE Brazil discussed above.



Iron plaques (dark blue) deposited on the external surface of mangrove roots

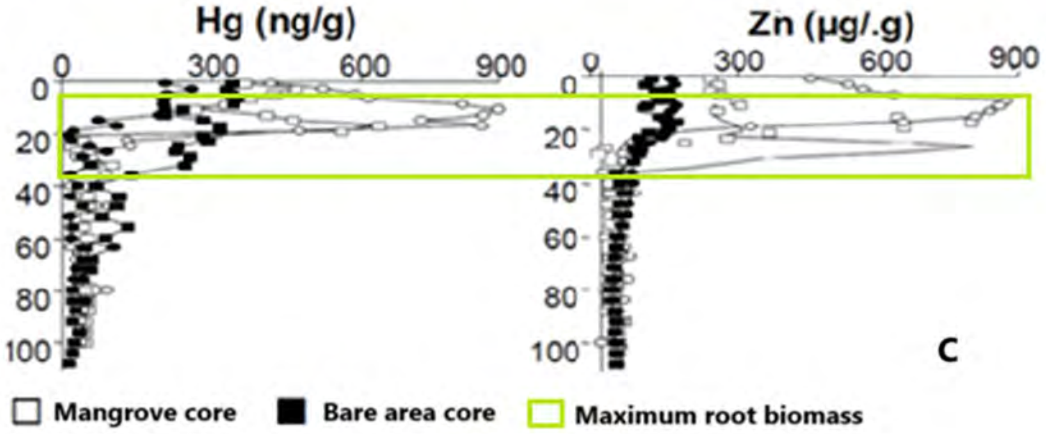


Figure 4. Mercury (Hg) and zinc (Zn) accumulation in mangrove soil cores compared to bare (degraded) areas, surrounding a landfill in Guanabara Bay, Brazil as iron plaques develop onto root surfaces (A). Adapted from Lacerda et al. (2000). Similar results for Cu in NE Brazil, showing the higher percentage of the total Cu concentrations associated with iron plaques (B) (Lacerda et al., 2024).

Unfortunately, the two sub-systems presented above, while very effective in immobilizing and accumulating metals, are biologically mediated and therefore can be disrupted by impacts on plant metabolism or on the activity of microorganisms. For example, due to hyper salinity in semi-arid regions resulting from reduced rainfall and increased saline intrusion (conditions typical of these semi-arid mangroves), the formation of iron plaques can be negatively affected, reducing the immobilization capacity in the rhizosphere. Similar to what was observed in the case of nutrients, degraded canopy health, and thus photosynthetic activity, may reduce ROL and the oxidation of phytotoxins. Environmental impacts associated with changes in hydrology and the sedimentation-erosion equilibrium may also impact the mobility of the deposited and accumulated metals. These changes are, however, very site specific, and will not be detailed here. On the other hand, environmental impacts caused by climate change are affecting mangrove ecosystems and their services worldwide (Lacerda et al., 2022). Thus, their effects on the pollutant immobilization capacity of mangroves pollutant immobilization capacity will be further discussed here.

2. How climate change driven pressures impact pollutant biogeochemistry in mangroves

Earlier studies on element cycling and balance in mangroves have shown that hydrology, mostly fluvial and tidal flux and the extent and duration of flooding, is the primary driver controlling the bulk fluxes of substances, including trace elements, nutrients, dissolved gases and alkalinity, from waterways to mangroves and from mangrove-dominated estuaries to the adjacent coastal waters

(Lacerda et al., 1988, 2020; Ovalle et al., 1990; Rezende et al., 1990; Kjerfve et al., 1999; Marins et al., 2020; Chielle et al., 2023a,b).

Increasing trace element concentrations in mangrove sediment cores in the Caravelas estuary, a large mangrove stretch in SE Brazil, were associated primarily with the eucalyptus monoculture, which enhanced erosion and runoff of contaminated soils from the river basin upstream (Angeli et al., 2018). In the semi-arid coast of NE Brazil, a greater flux of total Hg to mangroves was noted to occur during intense rainfall periods, associated with increasing loads of suspended sediments (Lacerda et al., 2013). Similarly, large nutrient and alkalinity fluxes also correlate with water flow (Ovalle et al., 1990; Marins et al., 2020) in mangroves of SE Brazil. Therefore, because runoff, fluvial flows and ocean forcing are intimately associated with climate change, it strongly affects the quantity of trace-metal loads to mangroves and thereon to estuaries and coastal areas.

The impacts of climate change on the hydrology of coastal areas are being observed globally (Dai et al., 2009) and the effects on the balance of pollutants in mangroves is significant and influenced by sea level rise. Seasonal monitoring of trace metal fluxes in mangrove-dominated estuaries revealed significant increases following extreme rainfall events (Lacerda et al., 2013). In addition, the seasonal variation of trace metal concentrations and speciation in shelf waters adjacent to mangrove coasts also suggest an increase in total trace metal fluxes following increasing river fluxes (Lacerda and Marcovecchio, 2018; Lacerda et al., 2020). Climate change has also been noted to concentrate rainfall periods (Andrade et al., 2018; Alvalá et al., 2019). This likely results in pulses of high trace metal fluxes to the ocean. These fluxes would be predominantly composed of particulate fractions of trace metals, which will most likely be deposited in continental shelf bottom sediments. In addition, extreme ocean and continental events will affect the erosion-sedimentation equilibrium and will potentially remobilize the historically accumulated metals from mangrove sediments.

Although the magnitude of pollutants' fluxes is largely controlled by hydrology, their chemical partitioning and speciation, and therefore bioavailability and toxicity, will depend on the interaction between multiple stressors of local, regional and global origins, that interact with the peculiar biogeochemistry of mangrove ecosystems. Hydrological variables, including residence time of water in the estuary, sedimentation rates and the redox state of surface and pore water are the major factors controlling the chemical speciation of metals and by extension, their mobility, bioavailability and toxicity.

Alterations in sediment dynamics and sedimentology due to changes in land-use and rates of weathering also influence the chemistry of metals in water, pore water and sediments in mangrove-dominated estuaries. Finally, pressures from changing tree metabolism, mostly at the root level, affect rhizosphere characteristics and pore water chemistry, also potentially affecting trace metal behaviour in mangrove sediments.

Table 2 summarizes the major drivers and their impacts on mangrove ecosystems and on trace metal biogeochemistry. All drivers impact the fate of pollutants in mangroves either through changes in hydrology and/or the sedimentation-erosion equilibrium. Also, their effects are similar, and generally result in increasing mobilization and bioavailability. Therefore, in real situations, it is generally very difficult to pinpoint a single or even the most important driver of a given change, meaning that the overall impact on metal biogeochemistry in mangroves is an integration of all drivers of impacts derived from climate change.

Table 2. Drivers of impacts and their effects on mangrove ecosystem functioning to mitigate pollution associated with global climate change.

Drivers	Impacts on the ecosystem	Effects on the biogeochemistry of pollutants in mangroves
Sea level rise	Erosion at the sea margin, tree felling and sediment resuspension. Landward migration following increasing saline intrusion and tidal forcing. Remobilization and oxidation of bottom sediments. Salinization of porewaters.	Oxidation of sediments leads to iron phosphate precipitation decreasing nutrient availability of plants. Oxidation and dissolution of sulphides, dissociation of metal-sulphides and release of associated chalcophile metals, facilitating metal-chloride complex formation (mostly Cd and Pb). Expansion of sulphate reduction upriver.
Annual rainfall reduction	Less continental runoff, exacerbating the effects of sea level rise and inducing hyper salinity.	Increasing environmental stress on plant metabolism, reducing the immobilization capacity of mangrove rhizosphere by decreasing ROL and iron-plaque formation.
Increasing the frequency of extreme events	Mangrove dieback following storm events. Decreased carbon accumulation in sediments. Erosion at the sea margin, tree felling and sediment resuspension.	Increases export of suspended sediments to coastal waters, affecting primary productivity. Sediment oxidation and release of metals complexed with sulphide. Immobilization of P, decreasing plant uptake.
	Extreme droughts cause higher sedimentation and water residence time.	Increases metal reactivity. Increasing export of organometallic compounds (e.g. methyl-Hg) and organo-metallic complexes, particularly of Hg, increasing metal bioavailability to the biota.
	Floods increase export of suspended particles to the continental shelf.	Increases particulate metals transport to the continental shelf and deposition in shelf sediments.
Global warming	Poleward migration of mangroves and substitution of seasonal saltmarsh communities. Increasing plant metabolism.	Increasing absorption of metals by mangrove plants. Expansion of the perennial sulphate reduction metabolism and chalcophile metals accumulation in sediments in seasonal saltmarsh areas.
Buildup of atmospheric CO_2	Increasing forest productivity and litterfall production, fuels microbial metabolism, including sulphate reducing bacteria. Increase in the formation of iron plaques in the rhizosphere.	Increasing rates of organo-metallic complex production and export, increasing the accumulation of chalcophile metals. Increased trace metal accumulation in mangrove rhizosphere.
Acidification	Dissolution of carbonates, increase elements solubility.	Increases in the solubility of trace metals, dissolved metal carbonates, increases metal bioavailability.

3. Concluding remarks

This short review discusses the applicability of mangroves as a sink for pollutants that is derived from the peculiar biogeochemistry of these ecosystems and the physiological and morphological characteristics of their component tree species. While many experiments that have attempted to regenerate/rehabilitate/preserve mangroves have been relatively successful, increasing human pressure from local and global drivers may significantly and negatively affect mangroves and the important ecological service they provide. Although most pressures are site-specific, intensive shrimp aquaculture and climate change are affecting this mangrove service in modulating pollutant transfer at the continent-ocean interface on a global scale. Excessive nutrient emissions from shrimp farming decreases mangrove health and its capacity to immobilize nutrients, particularly phosphorus. Degrading mangrove health also affects the capacity of trees for ROL, and consequently their ability to immobilize metals. Climate change-driven pressures alter the geochemical equilibrium in mangrove soils. Pressures associated with changing hydrology from decreasing annual rainfall, augmenting the frequency and intensity of extreme climatic events, and increasing ocean forcing and sea level have affected mangrove functioning and distribution. As well, these processes have impacted trace metal biogeochemistry, generally releasing accumulated metals to porewaters, mobilizing soluble forms of metals and increasing metal bioavailability to mangrove plants and associated animals. Further research into the environmental and socio-economic impacts of losing this important ecosystem service should be undertaken in order to better develop pollution mitigation plans using mangroves as natural filters and sinks of pollutants.

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Chapter 5

Interaction of blue carbon and trace metals in a mangrove environment

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1. Introduction

This chapter introduces organic matter in sediments, and its relationship with trace metals in mangrove systems. After a description of the mechanisms and processes of accumulation, scenarios are proposed on the different changes or impacts that anthropogenic pressure and global climate change may have on the capacity of estuarine mangroves to store metal contaminants.

1.1. The carbon cycle and organic matter

First, it is interesting to look at what we are talking about when we talk about carbon. Indeed, there can be several meanings behind the word carbon in the mangrove context: blue carbon, carbon stock, carbon dioxide or even black carbon or sometimes organic matter. Among the different forms of the element carbon there is inorganic carbon, which is mainly made up of carbon in its oxidized form such as carbon dioxide (CO_2), the carbonate form or elemental forms such as graphite or diamond. Carbonates are themselves a particular form of CO_2, which in the presence of water transforms into carbonic acid, $CO_2 + H_2O \rightarrow CO_2$, then into hydrogen carbonate and carbonate depending on the pH. These forms of inorganic carbon are related to various environments and physical states; gaseous with the formation of CO_2 (the dominant form in the atmosphere), liquid, with the formation of carbonates (HCO_3^- and CO_3^{2-}) and also solid, when in the sedimentary environment, during the formation of calcium or magnesium carbonate, most often. Calcium carbonate is present in limestone rocks but also in the oceans where it will generally precipitate to the bottom sediments. Calcium carbonate or limestone is one of the main forms of carbon on the planet. It is also the form most sensitive to pH because, depending on the acidity, it will be found in gaseous form in the atmosphere or in solid form in rocks.

Another part of carbon is in reduced form, this is what is generally called organic carbon. It is represented by simple organic molecules whose formula are known that come from abiotic (redox, photoreaction, etc.) or biotic reactions from plants and animals. Carbon can also be integrated into the building blocks of living bodies, i.e., biomolecules, which can be complexed to highly complex structures (proteins, cell walls, lignins, exudates, etc.). This organic matter (biomass) is gradually degraded by microorganisms or by abiotic chemical reactions, including photochemical reactions, in the form of complex molecules of undefined formulae called geopolymers, which in fact include what is known as natural organic matter. These are numerous high molecular weight molecules whose structure are not elucidated, nor are they even elucidable. These geopolymers are often operationally distinguished by their separation protocol, i.e., humic acids, fulvic acids and humins. These three groups are separated by a pH-related extraction protocol, with fulvic acids being soluble at any pH, humic acids only at pH 12, and humins insoluble at any pH. Within the organic carbon category that can now be found in nature, this can be called synthetic organic carbon. These are industrial products such as plastics or synthetic molecules.

All these different forms of organic and inorganic carbon constitute the carbon pool of the planet and circulate among the different reservoirs that are the solid reservoir (rocks, sediments, soil, etc.), liquid reservoirs (ocean, lakes, rivers, etc.) and gaseous reservoirs (atmosphere). Whether in soils or in water, it is important to understand that geopolymers, i.e. degraded organic matter, represent approximately 30 to 60% of the carbon present in these environments. This part is often called refractory carbon because it remains in the environment for a certain time without undergoing degradation, or very slow degradation. Despite the controversies that may exist regarding the existence or not of these humic substances (Lehmann and Kleber, 2015), it is a fact that some organic carbon or organic matter takes much longer to degrade than simple molecules or certain fragile molecules.

Most elemental carbon is found in sediments and sedimentary rocks, the latter having stored it over millions of years, as sedimentary carbonates which are found stabilized in the earth's crust at the bottom of the oceans or in limestone rocks, rocks that are gradually leached. This compartment accounts for 20,000,000 Pg (petagram, i.e., 10^{15} g) of carbon on the planet (Bolin, 1977; Stevenson, 1994). The second major reservoir of organic carbon is dissolved organic carbon in the oceans, both inorganic in the form of carbonate and organic in the form of dissolved organic carbon, and represents approximately 40,000 Pg of C. The reservoir composed of biomass and other surfaces such as soils represents only 800 Pg of C, i.e. only a small fraction of the amount of carbon present on the planet (<0.1%). Finally, the atmosphere contains or represents approximately 700 Pg of C. Thus, the amount of carbon present in the atmosphere and the amount of carbon present in the form of biomass on the planet is approximately equal.

The relationship between the different reservoirs, i.e. the lithosphere, the pedosphere (organic carbon located in soils), the hydrosphere (organic carbon located in waters) and the atmosphere, explains why it is important to maintain carbon in stable forms. Exchanges between the lithosphere and the hydrosphere or the pedosphere are significantly slower than those between the hydrosphere and the pedosphere and the atmosphere. Thus, when there is an increase in the concentration of carbon in the atmosphere in the form of CO_2, in the hydrosphere or in the pedosphere, it takes a significantly longer time to modify or store the quantity of carbon, and even more time to integrate it into the lithosphere (Sigg et al., 2006). For example, the 4 per 1,000 initiatives, which consist of increasing the quantity of carbon in soils by 3 to 4%, requires a much longer time frame than a simple one-off decision or action. Because of these differences in kinetics, the capacity of the lithosphere (*via* the precipitation of carbonates in the oceans) or the pedosphere (*via* photosynthesis mainly) to reduce or reabsorb carbon from the atmosphere is limited in time. This limited capacity along with the concentration increases, in C in the atmosphere give rise to, among other things, to the greenhouse effect and the corresponding increase in temperature.

Organic carbon is therefore one of the important links between the atmosphere and the pedosphere. It is ubiquitous and is found in most environmental compartments. If we look at the biomass on the planet, it is mainly located in tropical areas where there is a high density of forest with less located in the northern latitudes. On the other hand, if we observe the carbon present in the soils, that is to say a stock of carbon that is a little more refractory, the latter mainly occurs in the northern and southern latitudes and is much less prevalent in the equatorial zones. This is due to microbial activity in the soils, which is much greater in hot zones and which, therefore, consumes the organic carbon in the soils, hence producing CO_2. Thus, determining the quantity of carbon in the soils in isolation is not sufficient. It is also necessary to consider the general dynamics of the soils and vegetation.

Among the environments that concentrate a large amount of carbon within them, mangroves are very efficient carbon sinks. They can contain 3 to 5 times more carbon per hectare than tropical forests. Although representing 0.5% of coastal areas, mangroves contribute approximately 10% of carbon exported to the oceans, so they are very dynamic systems that play a key role in the carbon cycle. This is why it is necessary to study them. But, just like soils and forests, mangroves are also heterogeneous depending on geographical location. Located mainly in the equatorial belt, their productivity differs depending on the continent (Jardine and Siikamäki, 2014). The mangroves of the Indo-Pacific and Central America are significantly more productive in terms of the amount of carbon per hectare than those of the Atlantic coast. On average, a mangrove soil can contain up to 369 mg $C.ha^{-1}$ over the first meter, which is the depth layer generally studied, with few studies considering depths beyond one meter.

It has been shown that the most efficient mangroves for carbon storage are the Indo-Pacific mangroves ahead of those in the Atlantic. Generally, most are below the average of 239 g $C.m^{-2}.yr^{-1}$ (MacKenzie et al., 2021). It is agreed that the type of mangrove, i.e. its morphology, also plays a role in the quality or quantity of carbon storage. In general, riverine mangroves and tidal coastal mangroves are the most efficient at storing carbon, followed by lagoon-type mangroves, and finally those mangroves growing on carbonate terraces. Another factor that also comes into play regarding the storage capacity of mangroves is the age of the ecosystem. Young mangroves represent carbon production and a low storage capacity with an increase at around 10 to 20 years of age followed by a decrease in this storage capacity. Thus, mangrove restoration, for example, should not be considered as just a simple action without significant environmental impacts; long-term studies must be carried out to ensure the relevance of the action.

1.2. What are the dynamics of organic matter and metals in a mangrove system?

Dissolved organic carbon that arrives in estuarine mangroves *via* rivers encounters sea water whose geochemical properties are clearly different. The pH, chemical composition and salinity change abruptly, and the organic matter undergoes conformational changes forming particles or colloids. These particles deposit in the middle of the mangrove and feed a sediment that will contain a large quantity of labile organic matter. This process, which has been described by many works (MacKenzie et al., 2021; Mcleod et al., 2011) makes mangroves to be buffer zones between the terrestrial environment and the coastal zone. The mixing zone where sea water and river water meet depends heavily on the tide but also on river flow, which means that mangrove areas will spread out a distance both upstream and downstream where the salinities and pH conditions allow the development of mangrove forests. The ebb and flow of tides ensures that in the mangrove sediments, there are strong daily dynamics, with numerous transformation reactions, precipitation and sequestration. In addition, the vegetation supplies native organic matter through its leaves, fruits and roots, which degrade within the mangrove.

As for the metals, they will undergo the same geochemical transformations linked to the meeting of sea water and river water, and their chemistry will ensure that they will also precipitate in mangrove systems. Certain metals will precipitate directly by their chemical transformation into an insoluble complex such as, for example, carbonates or chlorides. Certain metals will associate with organic matter, that is in the process of flocculating either by complexation or by absorption and thus accompany the organic matter into the mangrove sediment. This process plays a buffering role in terms of contaminants. The alternation of tides also generates a strong dynamic within the sediments, and can change (daily), the speciation of metals, which can then either be stored in the anoxic sediments in the form of sulphides or be exported little by little with regular oxygenation of the surface sediments.

Particulate organic carbon comes mainly from biomass, but also from particles brought by the river. When the speed of water changes in the meanders of a mangrove, particles will deposit and then contribute to the formation of sediments. Particulate metals, associated with these particles, will also join the sedimentary system of the mangroves. All this organic matter in the sediments serves to provide food or energy to the microorganisms in the sediments, which will generate a redox gradient from the surface to the depths in the sediments, causing diagenesis which stratifies the chemistry of the contaminants throughout the sediment depth. The organic matter will therefore be degraded very quickly at the surface in the presence of oxygen. When there is no longer any oxidant such as oxygen, iron or manganese oxides, the degradation will continue slowly with the aid of specific microorganisms adapted to an anoxic environment. This organic matter will be more or less stabilized because degradation in the absence of the oxidant is significantly more difficult and energy-consuming. The contaminating metals associated with this organic matter will little by little undergo exchanges between dissolved forms or insoluble forms; they will also accumulate on the particles in anoxic environments, thus stabilizing them. This is how mangroves play a buffering role for trace metals.

2. Stock of metals in sediment

The stock of trace metals in a mangrove system was studied by Silva et al. (1990; 2006), who showed that this distribution also depends on the metal, some metals being more easily transported through the biomass from the root to the leaf, while others remain in the sediments. For example, iron and lead remain mainly in the root environments or in the sediments, whereas copper and aluminum are found mainly in the roots, while zinc, nickel and chromium are found in the biomass of the trunk. Cadmium for its part is capable of being found both in the sediments and in branches. A portion of these contaminants can then be transported from the sediment to the aerial part of the plant (Bourgeois et al., 2020). When the leaves fall on the sediments, some of the metal returns to the surface of the sediment is released again during the degradation of the biopolymer parts of the leaves and a new cycle can be set up. Regarding the import-export balance of these metals, meaning what quantity is exported by the mangrove and what quantity is stored by the biomass, it can be seen that iron, nickel and mainly aluminum and to a lesser extent lead are immobilized and stored by the mangrove biomass system. For certain other elements, such as Cu, Cd or Cr this storage effect is a little less obvious. The question therefore arises as to whether the mangrove is really a net sink for contaminants or if it can become a source of metallic contaminants.

2.1. Estimation of metal stocks

It is important to note that in the literature, there is no systematic evaluation of metal stocks in mangroves and so it is difficult to find comparable data. Figure 1 shows a non-exhaustive representation of the stocks of metals (kg.km^{-2}) found in mangrove sediments summarizing results from different studies worldwide (Szafranski and Granek, 2023; Kulkarni et al., 2018b; Marchand et al., 2006; Lacerda et al., 2001; Gaillardet et al., 1997; Harbison, 1986a; Jonathan et al., 2010; Uddin et al., 2019; Bastakoti et al., 2018; 2019; Ray et al., 2006; Ma et al., 2024). The scale is logarithmic, and we see at first glance that on average, mangroves contain higher stocks of Cu, Fe, Pb and Zn. To a lesser extent certain contaminants such as As, Mn and Cr are present in certain mangroves in certain countries. Thus, to better understand the risk associated with the potential release of these metals in coastal areas, it will be important to measure the metal content in different mangrove systems more precisely in order to have a correct continental-scale, even a worldwide balance.

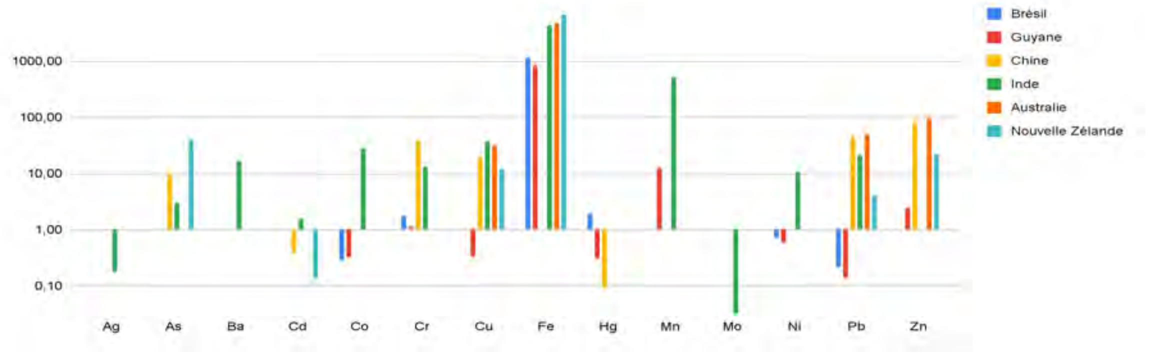

Figure 1. Average metal stock estimates (kg.km^{-2}), in mangroves from different countries.

After more observations, the role played by organic matter should be questioned. Generally speaking, the carbon concentration in mangrove sediment is high at the surface and decreases very quickly down to a depth of around 20 cm; the same is true for particulate and dissolved nitrogen and the decreases are also strongly linked to the presence of clay. The organic matter seems to follow the trend of clay with a high concentration at the surface and then, a rapid decrease linked to degradation. The quantity of carbon remaining at the bottom of the sediment represents carbon that is refractory to degradation under the dominant redox conditions (Cheriyan et al., 2022). When looking at the trace metal content in the sediments at the same time, it can be seen that the quantity of metal follows and/or is very well correlated to the quantity of carbon and the quantity of clay. An example is the regular increase in the trace metal content (Cu, Zn and Pb) corresponding to the regular increase in the quantity of organic matter as well as the quantity of clay in the sediments (Harbison, 1986). This, therefore, shows that metal content depends on the type of soil or sediment and the type of cover of the mangrove forest. Thus, both the heterogeneous nature of the contaminant content and the carbon content of mangroves make it difficult to simply multiply a surface area by a concentration. It is necessary to know in detail the composition of the mangrove (Harbison, 1986).

Likewise, activities upstream of a mangrove represent an important parameter contributing to the contaminant content of the sediments. These systems being dynamic, the change in upstream activity also changes the metal concentrations, demonstrating that these dynamic systems are not just sinks. Several studies clearly show that during the dry period when the use of mangroves in Gandon province, China, transitioned from an industrial economic activity to a tourism activity, there was an increase in the quantity of Cr in the sediments, as well as an increase in the quantity of Pb (Hasan et al., 2023; Uddin et al., 2019). At the same time, there is a decrease in the quantity of Pb and Zn and an increase, despite everything, in the amount of Cr. The decrease in the quantity of Pb and Zn suggests these metals were exported to the coastal zone meaning that the buffer role of the mangrove did not take place.

3. Is organic matter only energy for microorganisms in sediments?

It has consistently been shown that organic matter has a complexing power with respect to trace metals. Trace metals are associated with a certain quality of organic matter that facilitates the transport of the latter by complexing them, that is to say, by changing their chemical nature (Lu et al., 2019; Thanh-Nho et al., 2020). In solution, metals can have several chemical forms. They can be associated (as described before) with particles by being either absorbed or complexed by chemical sites, which form solid forms containing metal. Metals can also be complexed with dissolved inorganic species (Cl^-, OH^-, SO_4^{2-}, CO_3^{2-}, etc.) or with dissolved organic molecules, which give them slightly different properties in terms of bioavailability and toxicity. In fact, the organic complexes of metals are less toxic than the free metal. Finally, the free form of the metal, that is the metallic cation without any molecules around it (except water molecules), is generally the most toxic species. All these metal species constitute metal speciation, i.e., the distribution of the metal element among the chemical species. The ability of metal to be translocated from an inert to living medium through root cells depends on the amount of metal in a particular form. In general, it is the free form that is taken up; however, certain plants or certain cells produce specific complexes to allow the necessary quantity of metal (for their metabolism) to be absorbed in a controlled manner. In some cases, the exudates serve to eliminate the toxic metal by metabolic transport in both complex and inert forms.

The metal species that are transported during the liquid-solid exchanges that take place during the tides are mainly the free metal and the dissolved complex molecules. It is therefore important to know the way the metal is transported, and/or to predict its speciation, if we want to know its fate or its future in the environment. There are several ways to determine the quantity of metal that is likely to be exported or likely to be absorbed by plants.

3.1. Acid extractable determination

One of the ways of estimating these chemical species is to determine the acid-extractable fraction. The sediment sample is put into an acidic medium and the quantity of metal that is extracted is measured. A weak acid is generally used to simulate the weak extraction by organisms. This methodological fraction provides information on, for example, the desorption of complexed metals and/or their association with carbonates (since the latter disintegrates) but does not predict metals speciation. This acid-extractable fraction can have correlations with biological indices such as the Effect Range Low (ERL) water and can also furnish recommendations on sediment quality (Jingchun et al., 2010).

3.2. Diffusive gradients in thin films

Another way of understanding the metal contents of the environment is by use of Diffusive Gradients in Thin (DGT) films. This technique uses a resin with a high complexation capacity that is separated from the medium by a gel; the diffusion of free metals through the gel allows an integration of the quantity of free metal present in the system. Thus, the quantity absorbed by the resin corresponds to the quantity of free metal perceived by the medium and therefore measures the amount of metal that is able to pass through the cell membrane. This methodology makes it possible to understand the quantity of free metal, which is very sensitive to concentration but also to temperature, and is the closest measurement of what we call the bioavailable metal fraction (Kastury et al., 2023).

3.3. Modeling speciation

Finally, a third way to predict metal speciation is through modeling. Modeling makes it possible to calculate the exact speciation of metals; provided the complexation constants and the concentrations of ligands in the environment are known. In general, the thermodynamic constants of inorganic species are known, however it is difficult to obtain (without specific experimentation), the constants between organic ligands and metals. Determining these constants, however, would allow us to have information on all the metal species. A global vision of the environment makes it possible to ascertain which species are present and which of those are toxic and also makes it possible to carry out modeling under different conditions such as pH, redox or temperature. As well, these constants could be integrated into hydrodynamic models (Lützenkirchen et al., 2015).

3.4. Modeling nickel speciation during a tidal cycle

To illustrate the previous method, the following will present a complexation model developed over 24 hours in a mangrove in New Caledonia. During this experiment, 24 samples were taken using an auto-sampler; salinity, pH and temperature were also determined. Figure 2 shows the salinity and pH of the Temala mangrove during the experiment which was studied during a TREMOR project in 2019 in New Caledonia (2024.01.07: https://cresica.nc/projet/tremor).

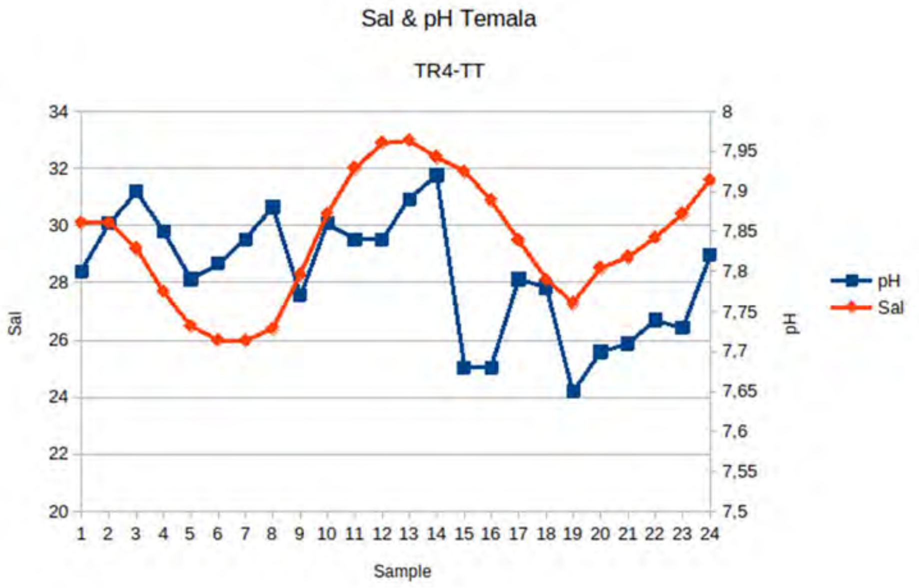

Figure 2. Salinity and pH of the Temala mangrove site during the 2019 TREMOR project

The samples were analyzed using fluorescence quenching which, using a metal titration, permits the determination of complexation constants between the organic matter present in the solution and a metal (Ryan and Weber, 1982). In this case the added metal was nickel. During quenching, each sample was analyzed by Excitation Emission Matrices (EEM) fluorescence, called 3D fluorescence, and each matrix was decomposed using a PARAFAC algorithm (Stedmon and Bro, 2008). Two groups of fluorophores, a humic-like fluorophore and a protein-like fluorophore were measured. These two compounds were analyzed by fluorescence and show variable concentrations over time (Figure 3).

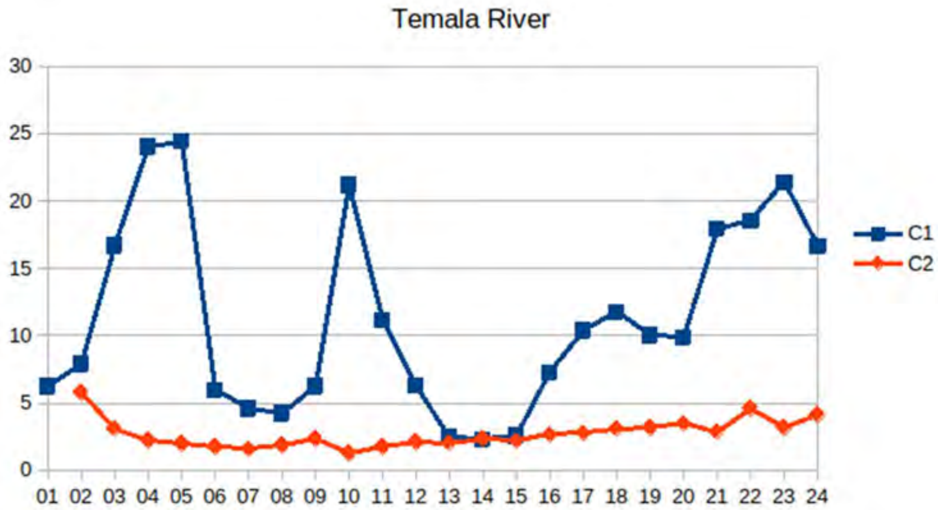

Figure 3. Time evolution of the humic-like (C1) and protein-like (C2) metal fractions during a 24-hour tide sampling.

Each sample underwent a fluorescence quenching experiment, and the modeling results allow extraction of the complexing properties of the components. During the 24-hours of measurement, it appears that during the night some samples showed strong complexation constants with Ni (Figure 4a). This means that the organic matter in the mangroves has a significant daily dynamic, and that Ni complexation, that is its speciation, will vary over time (Figures 4b). The maximum complexation occurs when a lower quantity of Ni is present i.e., in the middle of the night at high salinity and high tide. Using its thermodynamic values, a speciation calculation was carried out taking into consideration the inorganic content of the elements in the solution and the Ni content of each solution. The result is that during the 24-hour cycle, part of the Ni is complexed with * organic matter, particularly during the night when almost 80% of Ni is in organic form compared to the rest of the time when Ni is present in solution with less than 10% in the organic form (Figure 4b). Thus, Ni speciation differs during the tide and is dependent on the tidal cycle and the organic matter present in solution. This experiment underlines the importance of organic matter in the speciation of metals and therefore their fate during significant hydrodynamic events.

The important role of organic matter in the speciation of metals was also reported by Taillardat et al. (2018) and Xiao et al. (2023). Their studies demonstrated that when the quantity of organic matter increases in export, the quantity of metal also increases. Organic matter therefore, augments metal export from the mangrove system which no longer act as a buffer for metallic contaminants. Copper and arsenic increase sharply when organic matter also increases, regardless of the different sampling seasons (Taillardat et al., 2018; Xiao et al., 2023).

By associating this hydrodynamic behavior with the stocks of organic matter and metals, it is possible to calculate the quantity imported or exported of certain metals, for example Cu, for which all measurements are available. Considering that the quantity of copper stored by the litter cycle is approximately 0.002 mol.ha^{-1}.yr^{-1}, while carbon flow results in an export of Cu from the mangrove of 0.15 mol.ha^{-1}.yr^{-1}, and considering the stock of copper in the sediments (~22 mg.kg^{-1}), this leads to a concentration of 3,600 mol Cu.ha^{-1} resulting in a net export of Cu of 0.113 mol.ha^{-1}.yr^{-1}. This export is not enormous compared to the stock of Cu but shows that the mangrove does not behave as a barrier to copper contamination but rather as a source of Cu contamination. Thus, depending on dynamics and the quantity/quality of the organic matter, mangroves may actually be sources of metal contaminants rather than stocks. What actually occurs is that due to sediment processes and the dynamics of organic matter, there is a transformation of the metal by plants and microorganisms which changes it from one form (arriving to the mangrove) and leaves by another form during the export of organic materials from the mangrove.

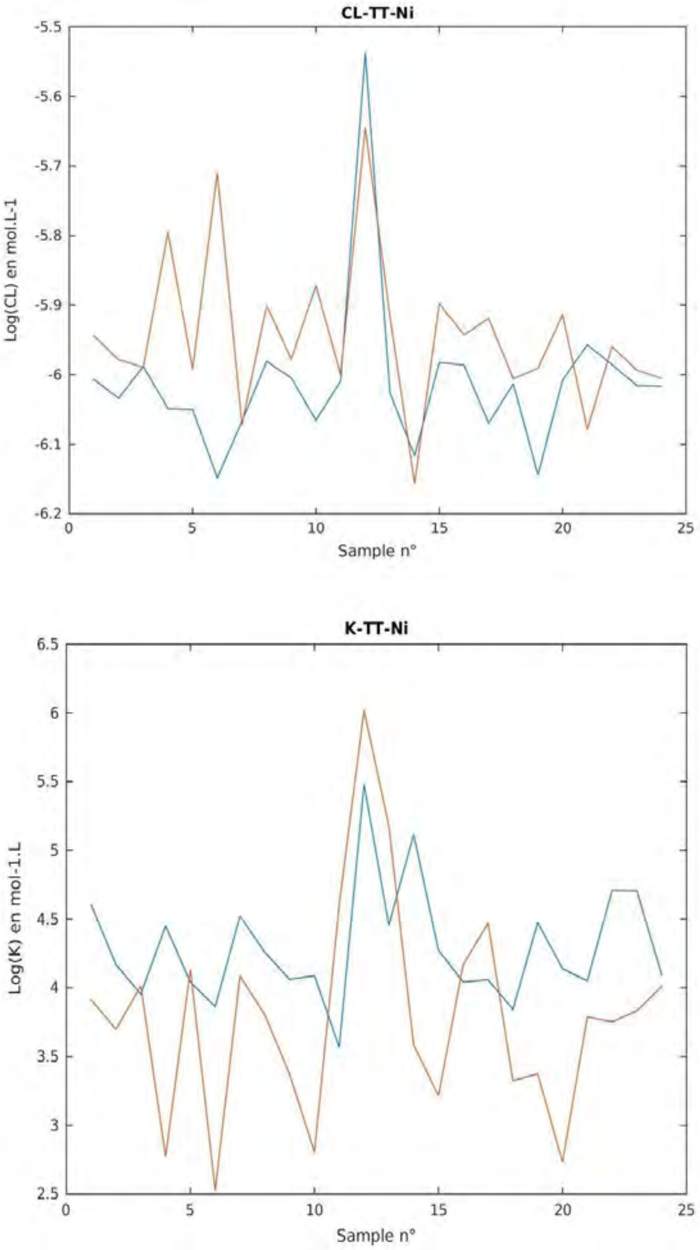

Figure 4a. Some night samples showed strong complexation constants with Ni.

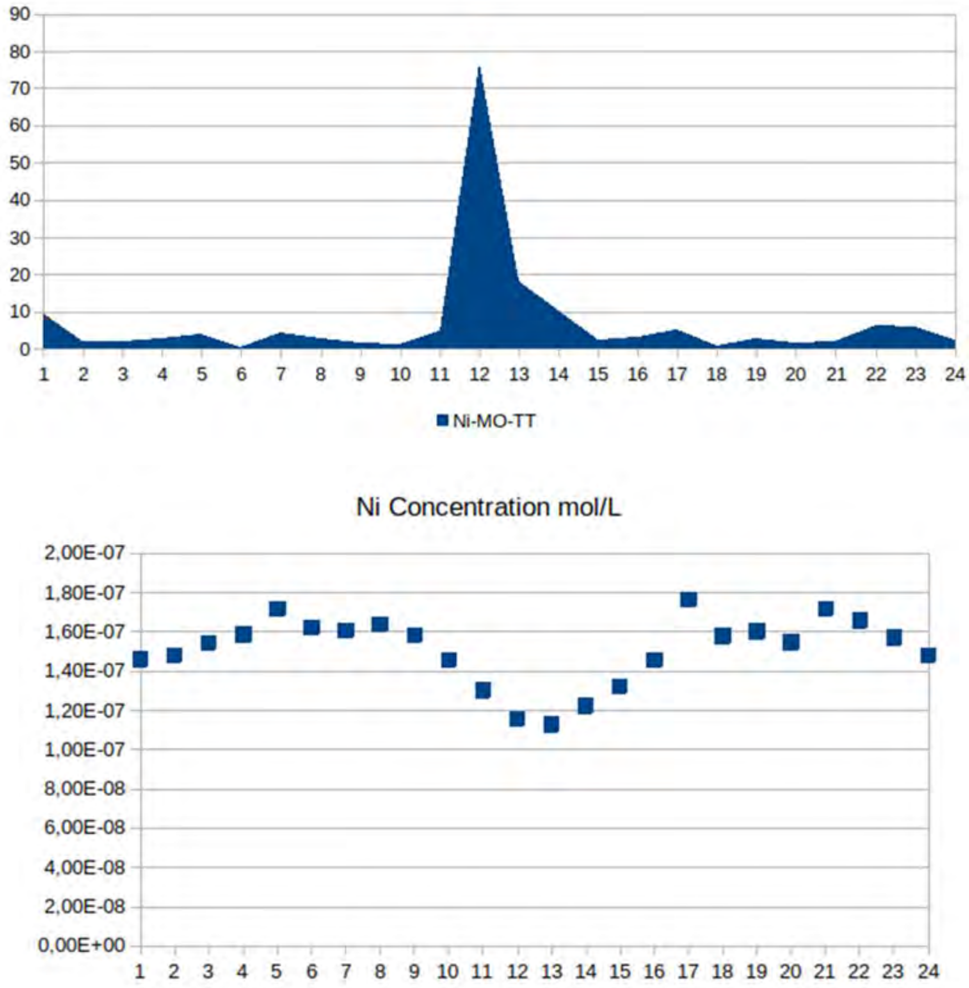

Figure 4b. The organic matter in the mangroves has a significant daily dynamic and that Ni complexation or speciation will vary over time.

4. Complexation constant and complexation capacity

Organic matter plays an important role in the export of trace metals in mangrove systems. However, what is the influence of climate change and anthropogenic pressures on this organic matter and therefore the consequences on the behavior of mangrove systems with respect to metallic contamination?

4.1. Urban pressure

Certain mangroves exposed to urban pressure are subject to nutrient inputs that increase bacterial activity and, consequently, decrease the amount of carbon in the sediments. A transformation in the quality of the organic matter in these mangroves also occurs (Cheriyan et al., 2022; Bourgeois et al., 2020). In the region of Fortaleza, Brazil, urban mangroves have a significantly weakened capacity for organic matter complexation compared to a mangrove located in a much more protected area. In fact, in the urban mangrove located in the Cocó River Estuary, the complexation constant, that is to say the ability to complex, is approximately the same as upstream mangroves; however, the number of complexing sites for organic matter is statistically lower in the urban Coco River mangroves than in the pristine Pacoti River mangroves (Mounier et al, 2018). This points out that an urban-impacted mangrove has a lower capacity to retain metals than if it had not been impacted.

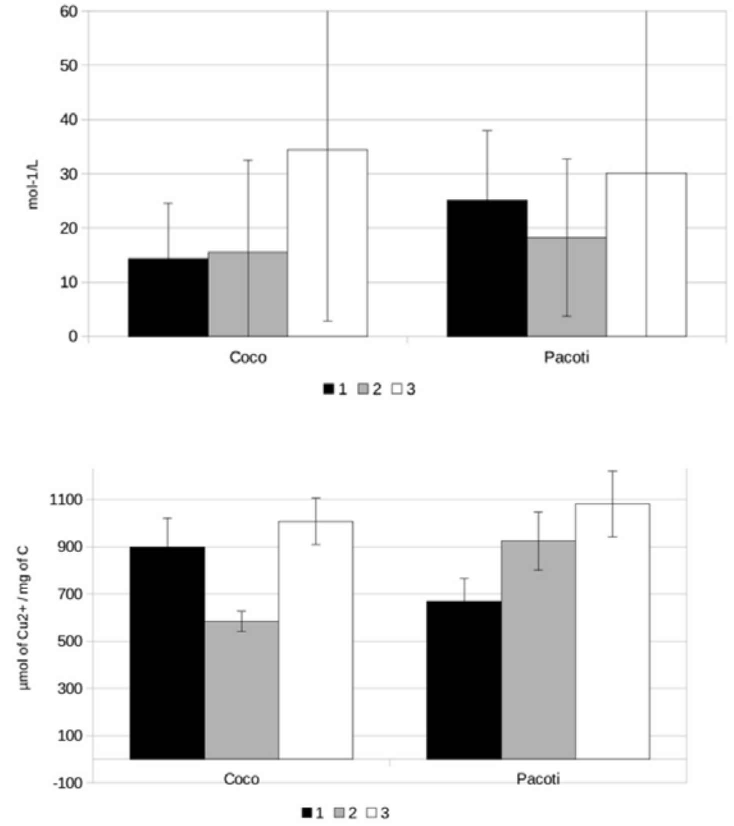

Figure 5. Complexing capacity (left) and complexation constant (right) of organic matter and copper at the Cocó River estuary (urbanized) and the Pacoti River estuary (pristine) from upstream and downstream of the mangrove forest (Mounier et al., 2018).

4.2. Frequency and tide amplitude

Another subtle factor which can cause a modification of the complexation capacities of a mangrove's organic matter is the frequency and amplitude of the tides. With the increase in sea level, it has been shown that the amplitude and frequency of tides can affect the organic matter in the sediments (Taillardat et al., 2018), resulting in greater microbiological activity and a more frequent supply of oxygen, thereby increasing the quantity of organic matter that will be transformed or digested by sediment microorganisms. When this organic matter is transformed, some of the metals associated with its surface or structure are released. These new free metals return to the environment and undergo, again, redox exchange and speciation changes. At this point, they can return to a passive state in the form of solids or particles; however, if this return is not fast enough, there may be an export of metal species in solution. Thus, tidal amplitude will modify the balance of diagenesis in the sediments, and more frequent erosion or oxygenation will also cause an increase in the quantity of metals released. At the same time, an increase in the transformation of organic matter into CO_2 during bacterial respiration will occur, thus contributing negatively to the storage balance of mangroves. The impact of these dynamics is very difficult to determine but it is important to recognize the potential carbon 'bomb' (reservoir) that constitutes refractory carbon in mangrove sediments. Extreme events also have an impact on mangroves because they destroy the coastline and transport anoxic sediments into oxic environments, again causing a transformation of organic matter and the release of metals. It has been clearly shown that when an extreme event occurs, the quantity of metals present in the surface sediments of the coastal zone increases drastically in front of mangrove areas, thus metals transported by rivers and sequestered by mangroves are exported to the coastal environments and mangroves once again find themselves a source of contaminants rather than a barrier (Gopal et al, 2017).

4.3. Rising temperature

With respect to global climate change, it has been shown that increasing temperature increases the reactivity of bacteria in the sediment and thus increases carbon export. This increase in carbon export coincides with a decrease in the amount of carbon in the sediments and is linked to greater bacterial activity in its sediments. An increase in temperature and rainfall increases mineralization in mangroves and thus the degradation of organic matter, which may contain metals that will therefore be released during degradation (Van Vinh et al., 2020; Jardine and Siikamäki, 2014; Zhang et al., 2021).

5. Conclusion and recommendations

In conclusion, it should be considered that organic matter in mangroves is a very dynamic system. Organic matter has a significant affinity with clays and constitutes a storage pool of carbon by becoming more and more refractory in soils. However, this stability can be an illusion in the face of future pressures from climate change and other activities. In particular, mangroves have significant sensitivity to anthropogenic pressures and the supply of nutrients, which can be introduced into a system and degrade the organic matter more effectively. It is indeed notable that urban mangroves have a significantly lower carbon concentration in their sediments than non-urban mangroves. Fresh organic matter carried into these systems increases respiration, decreases the carbon stock and of course, contributes to increased CO_2 in the atmosphere. Fresh organic matter also reduces the complexation properties of mangroves (due to the complexation by organic matter) or the complexation of metals by organic matter and therefore reduces their storage capacity. Finally, the transformation (or export to an oxic medium), of this refractory organic matter generates organic matter that is available to bacteria; therefore, the refractory nature is only an illusion as part of this organic matter or a part of this carbon stock may be released into the atmosphere through this new transformation. Thus, the degradation of organic matter in mangroves has a negative retroactive effect with regard to global climate change. Contrary to what one might think, the quality of organic matter in mangroves is highly sensitive to climatic conditions and anthropogenic pressures and must be closely monitored so as not to generate positive retroactive effects.

Similarly, metals are significantly linked to the quantity and quality of organic matter in the sediments. It is not only biomass that retains metals in mangroves, in fact, sedimentary diagenesis is also an important driver of metal stabilization. However, it is necessary to consider the fact that sediments are sensitive to reoxygenation linked either to anthropogenic activities or to the greater tidal amplitude, or coastal erosion. The fate of metals in mangroves is strongly linked to organic matter and clays, eutrophication causes a loss of the capacity of organic materials to stabilize metals, and of course the degradation of organic matter leads to a release of metals. Metals in mangroves are also very sensitive to global climate change and anthropogenic uses.

It is therefore recommended that more research be carried out to understand the lability of organic matter in mangrove sediments, and to understand how much organic matter is truly refractory in mangrove soils. It is also important to better characterize the quantity of metals in these sediment stocks and to study the losses in storage capacity of mangroves. Finally, detailed knowledge of the different types of mangroves, particularly in countries where there are few studies (e.g., the African continent), is warranted. All this work would allow us to better understand the effects of anthropogenic pressures and climate change on the capacity of mangroves to be sources or sinks of contaminants and carbon.

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Chapter 6

Rehabilitation and restoration of mangroves and the stocking of blue carbon

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1. The carbon cycle and mangroves

Carbon (C) is one of the most important nutrients, the main component of life on earth and the fuel of food chains. Carbon exists in many forms in the atmosphere, such as carbon dioxide (CO_2) and methane (CH_4), two greenhouse gases (GHGs) of high environmental significance. The Earth's atmosphere can trap heat from the sun, and despite being necessary for life on Earth, atmospheric heat has been uncontrollably rising due to the huge human use of energy and land, emitting GHG gases such as CO_2 and CH_4, as well as nitrous oxides and others.

Oceans are sinks for C, taking around 90 gigatons of C (GtC).y^{-1}, including one quarter of the anthropogenic CO_2 released into the atmosphere (Figure 1). One part of this C enters the phytoplankton photosynthetic processes, and by respiration and decomposition processes, returns to the atmosphere. Another part of the absorbed CO_2 (2 GtC.y^{-1}) is exported to the deep ocean. There, this 'biological pump' fuels a reservoir of C close to 50 times larger than the atmosphere. The ocean surface contains 1,000 GtC, while the deep oceans around 37,000 GtC, and their sediments, 6,000 GtC. On the other hand, the atmosphere contains 800 GtC. On land, total plant biomass contains 550 GtC, and plants take ~120 GtC.y^{-1} from the atmosphere, plus ~3 GtC.y^{-1} from human emissions. Soils lose 60 GtC.y^{-1} back to the atmosphere due to microbial decomposition and respiration, and from the continents ~9 GtC.y^{-1} are released from fossil fuel combustion, cement production, and land use changes. Approximately 10,000 GtC are stored as 'fossil' pools, and 1.4 GtC are transferred annually from land to oceans. Presently, the annual atmospheric increase in C is estimated to be 4 GtC (U.S. DOE, 2024).

Some coastal habitats such as sea grasses and tidal marshes, like salt marshes and mangrove forests, can play a significant role in regulating the Earth's climate, since they capture great amounts of CO_2 (so called blue carbon) from the atmosphere by photosynthesis, and trap it in vegetal tissues and in soils, accumulating a total mass of ~10-30 GtC and sequestering it for thousands of years, thus avoiding more atmospheric heating. Mangroves can fix a great amount of atmospheric C, around 3% of that being sequestered by tropical forests; notwithstanding these wetland coastal forests represent only 1% of the forest area of Earth. Yet, despite covering only 0.5% of the Earth's coastal area, mangroves contribute 10–15% (24 TgC.y^{-1}, i.e., 10^{12} gC.y^{-1}) of the global C storage in coastal sediments, while exporting 10–11% of the particulate terrestrial C to oceans (Alongi, 2014). Mangroves can stock these large amounts of C (more than 1,000 C.ha^{-1}) mostly in soils (49 to 98%) and the remaining in forest biomass (Donato et al., 2011).

All this accumulated C returns to the atmosphere when mangroves are cleared or degraded, generating proportionally high C emission levels. The amounts of mangrove stocked C are permanently updated and estimates suggest they are decreasing due to habitat conversion and degradation (Bryan-Brown et al., 2020; Goldberg et al., 2020). Still, mangroves offer more than this crucial service of capture and storage of C in the present climatic regime and provide other pivotal environmental services. They include fishery breeding grounds, nutrients fueling coastal productivity, biodiversity (and food) reserves, water cleaning and pollution abatement, coastal protection from climatic and marine extreme events like storms and waves, and awareness-raising human-nature interactions.

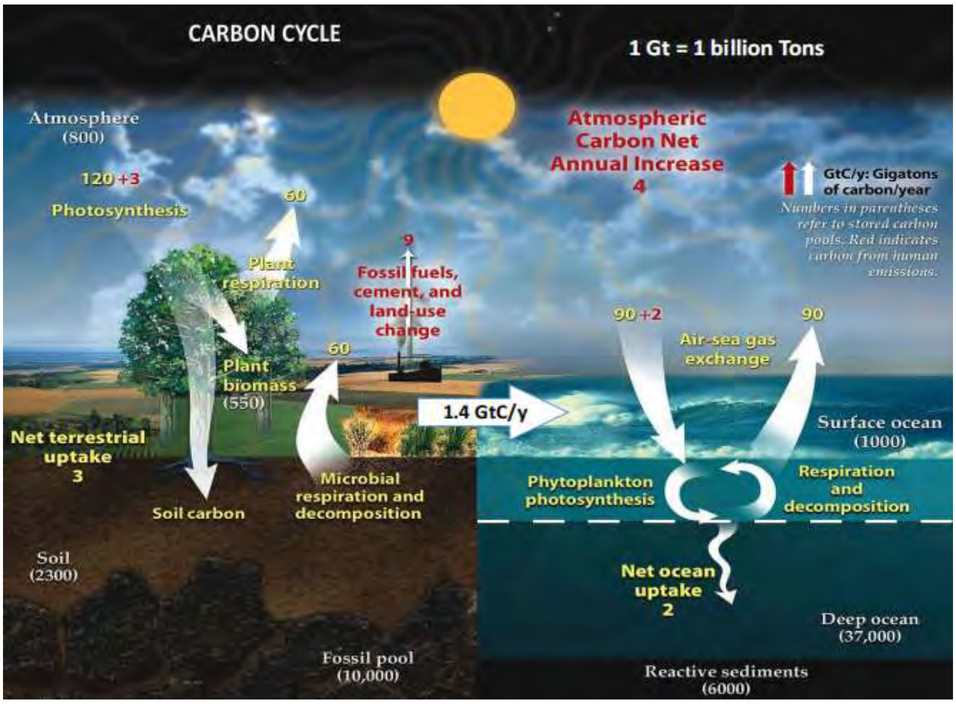

Figure 1. The carbon cycle (modified from U.S. DOE, Biological and Environmental Research Information System, at https://earthobservatory. nasa.gov, 2024).

2. Mangrove rehabilitation and restoration

2.1. Rationale

The extent of mangrove forests has been decreasing over the last decades (Bunting et al., 2022; FAO, 2023). Even though the annual percentage of mangroves degraded by humans has slowed over the last two decades, forests still are being suppressed for different reasons. The present extent of mangroves in tropical and subtropical coasts is around 13.67 (Worthington and Spalding, 2018) to 14.77 (Contessa et al., 2023) million hectares. Numbers are variable depending on measurement methods. Yet, besides human driven factors, climate changes are also directly and indirectly impacting these ecosystems, increasing physical and functional degradation (Ward et al., 2016; Ward and Lacerda, 2021, Ferreira et al., 2024).

The main mangrove drivers of degradation and their relative importance in the last two decades, from 2000 to 2010, and from 2010 to 2020, are i) intensive shrimp aquaculture, ii) extensive monoculture, agriculture and land use changes, iii) climate change, natural disasters and extreme events, and iv) direct and indirect effects from urbanization and industrialization (FAO, 2023). Table 1 shows the relative importance of each driver by the decade. Between 2000-2010, intensive aquaculture was the most significant degradation driver, higher than extensive monoculture, agriculture and land use change, together with climate change, natural disasters and extreme events. In the past decade (2010-2020) these two latter drivers were the most important causes of mangrove degradation worldwide, since the impact of intensive aquaculture decreased globally (Table 1), except for some countries in South America (Lacerda et al., 2019).

It is recognized that the better way to maintain the goods and services furnished by mangroves is to preserve them, avoiding their fragmentation and degradation (Friess et al., 2019; Bryan-Brown et al., 2020). Yet, restoring these forests where conversion or degradation has occurred is necessary and urgent, and possible if undertaken using tested methods, and following several steps to enhance success and decrease cost/benefit rates (Lewis, 2005, 2009; Ferreira et al., 2023). Several programs and projects have rehabilitated and recovered mangrove stands, from small to larger areas (Field 1998; Primavera and Esteban, 2008; López-Portillo et al., 2017; Ferreira et al., 2023). Studies on the progress of these restorations are valuable to learn and replicate. Presently, about 8,120 km^2 globally, of lost mangroves are restorable, while only circa 2,000 km^2 have been restored in the last 40 years, though undocumented restorations at several scales could increase this area (Worthington and Spalding, 2018).

Table 1. Relative percentage importance of main mangrove degradation drivers in last two decades (modified from FAO, 2023).

Driver	Relative importance 2000-2010 (%)	Relative importance 2010-2020 (%)	change (%)
1) Shrimp aquaculture	31.3	20.6	-10.7
2) Extensive monoculture, agriculture and land use change	24.8 palm oil 4.1 rice 12.3 other 8.4	34.5 13.6 3.3 17.6	+9.7 +9.5 -9.0 +9.2
3) Climate change, natural disasters including extreme events	~ 25	32.6	+7.6
4) Urbanization and industrialization (direct and indirect effects)	14.7	11.3	-3.4

2.2. Restoration definitions and protocols

Let's examine some terms and definitions to better understand mangrove rehabilitation /restoration (R/R). Frequently, we use terms such as 'rehabilitation', 'restoration' and 'recovery' indistinctly. Rehabilitation stresses on the reparation of some ecosystem processes, productivity and services, while the goals of restoration also include the re-establishment of the preexisting biotic integrity regarding species composition and community structure, according to standards of the Society for Ecological Restoration (SER, 2004). It is admittedly difficult to return to a preexisting state due to inherent permanent changes in natural ecosystems. More recently, the definition of ecological restoration was updated as the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed, implying that the degraded ecosystem be moved onto a trajectory of recovery but allowing its adaptation to local and global changes, as well as the persistence and evolution of its component species (Gann at al., 2019). This is an important definition, since it recognizes that environmental conditions and ecosystems change, and that the component species and their relationships are crucial to shape the community through ecological processes. Here, we will treat: i) restoration as a process, considering recovery as a result of this process, and understanding that rehabilitation can be a primary stage of the restoration process and ii) the biota as the most significant component in the process of mangrove recovery.

Several protocols and methodologies have been published, and countless mangrove rehabilitation/restoration projects and attempts (RRPAs) have been performed (Lewis and Brown, 2014; Teutli-Hernández et al., 2021; Ferreira et al., 2023) in both main geographic realms where mangroves exist, i.e., the Atlantic-Caribbean-East Pacific (ACEP) and the Indo-West Pacific (IWP) coasts. The IWP mangroves have several times more tree species (and other organisms) than the ACEP. Although not all studies have been published, most show that some RRPAs have been successful while others have not for various reasons. RRPAs are extremely variable, first in aims, targeted expansion (from small to large areas), tree species used and density, but also in post-restoration management. In general, the chances of success are inversely proportional to the area being extended. For small areas, it is possible to develop R/R attempts as experiments (adaptive management framework) (Ellison et al., 2020) and use them as case studies and/or more general models, to set new policies and protocols to guide further attempts. We briefly explain the R/R process based on a general framework designed by Bosire et al. (2008) (Figure 2) and a review of mangrove R/R protocols and practices by Ferreira et al. (2023).

The aims of mangrove R/R are variable, among them being the recovery of ecosystem services, the creation or maintenance of forest stands for biodiversity conservation, coastal protection, landscaping, legal requirements to repair damages, and others (Ellison et al., 2020). These goals need to be very clear before starting the R/R process since it is important to choose a proper methodology or experimental design (Gann et al., 2019; Ferreira et al., 2023). It is necessary to integrate subsistence and traditional mangrove users, learn traditions and perceptions of native populations and engage them for a sustainable use of mangroves. In fact, the RRPA can be designed to recover environmental goods and services lost by human populations (e.g. carbon sequestration, fisheries, timber production, clean waters, coastal protection, among others) (Field, 1998; Borges et al., 2017).

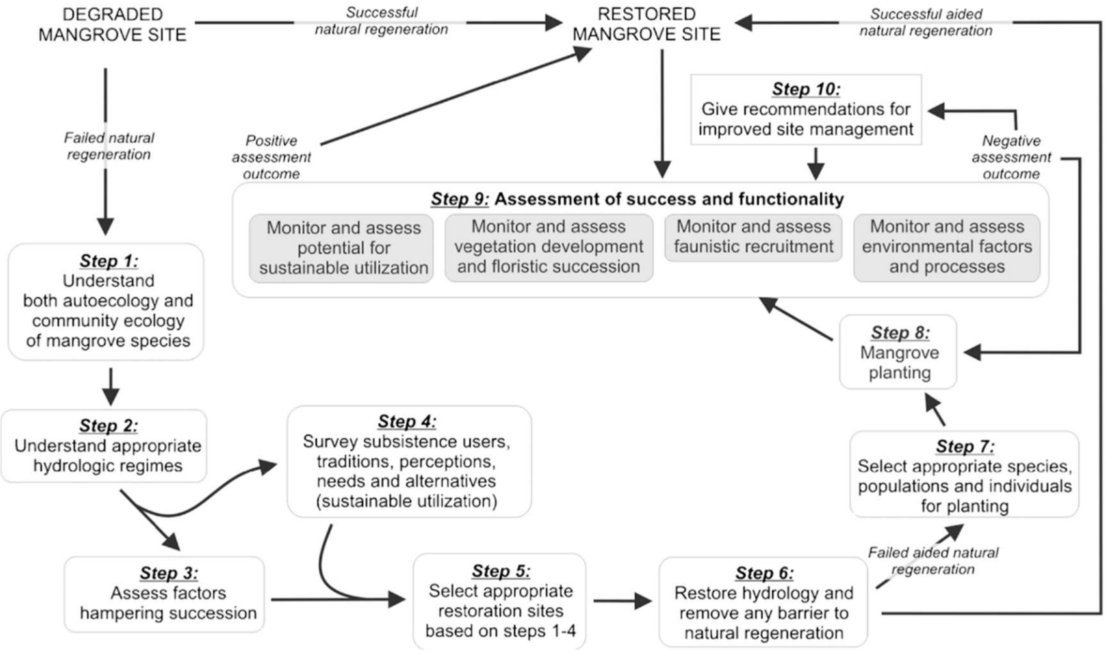

Figure 2. The rehabilitation/restoration steps (from Bosire et al, 2008), a general road-map of mangrove rehabilitation/restoration.

When a degraded mangrove is targeted for recovery, the first step is to understand why the forest is not naturally recovering by itself. For this, it is necessary to undertake a ‘forensic’ study, gathering information about the previous forest structure and degradation drivers of the area, their physical and biotic aspects, and if there are changes in topography, soil or hydrology at the time of, or after degradation (Teutli-Hernández et al., 2021). This is crucial, since it gives a comprehensive idea of the viability of the RRPA. If this information is lacking, it is necessary to identify the natural state of a regional reference mangrove relative to the physical-chemical environmental conditions (hydrological regimes and patterns, e.g., wet and dry season characteristics, tides, soil types, etc.), tree composition and their local distribution and response to natural or human induced environmental changes, on which to base the RRPA. Understanding the ecology and diversity of the extant tree species (e.g. life cycles, reproduction patterns, propagule features and establishing dynamics) (Lewis, 2005, 2009), is crucial to choosing the species and management type. Indeed, to further promote a more functional recovery, a knowledge of the biota is recommended to restore (mainly their functional groups) organisms important in the shaping of the physical substrate, forest architecture, community structure and food chains (Ferreira et al., 2023). The participation of users from traditional communities can provide valuable information on these issues and sustainable practices.

After that, it is necessary to remove any factors that could impair natural regeneration or secondary succession, for example, retention of propagules before reaching the R/R area, colonization and competition by terrestrial plants, cattle grazing/trampling the soil, and any physical barriers that may encroach on the area and/or impair mangrove settlement. The supply of propagules by tides is crucial to mangrove recovery, so they need to reach the targeted areas; in some cases, this is sufficient for mangrove recovery (Lewis, 2005; Bosire et al., 2008). This is one of the motives to restore or redesign hydrology, that is, to allow tidal or estuarine waters to reach the target areas, since they carry propagules and water with nutrients. Then, if hydrological restoration promotes the selected regeneration, the mangrove is on course for natural R/R.

The next crucial step is the monitoring and assessment of the rehabilitation process, addressing environmental factors and processes, like changes in biotic and abiotic features, forest development and floral succession, and the recovery of ecosystem functionality following the return of the biota, mainly functional groups involved in significant ecological processes, such as herbivore and fossorial crabs, pollinators, and specific soil microbiota, among others (Bosire et al., 2008; Ferreira et al., 2024). This monitoring step is historically one that is most needed, however, few assessments are performed, hampering the acquisition of valuable data about the RRPA; again this can be mitigated, somewhat, through a synergy of traditional and scientific collaboration and knowledge.

At this point, passive and active restoration diverges. If recovery was not successful naturally or assisted through hydrological recovery, the solution could be through planting. This is the time to select appropriate species, better cost-benefit density, and healthy propagules for planting. To obtain better recovery of mangrove ecosystem functions, the use of native species that grow in reference areas, is preferred (SER, 2004; Gann et al., 2019). This approach makes it difficult to recover species-rich mangroves, which are more functionally diverse. For example, in the IWP realm, some regional mangrove stands have up to 15 species, each likely with narrower specific ecological requirements in the littoral fringe. So, the IWP mangroves may not easily respond to R/R attempts to plant specific propagules/seedlings at a different littoral fringe level than required by the species (Ferreira et al., 2023).

Conversely, in mangroves with few tree species, such as in the Neotropics or in semi-arid and arid coasts, restoration can be more promptly reached, since one or few tree species can constitute the full set of extant species (Toledo et al., 2001, Al-Khayat et al., 2019; Ferreira et al., 2023). That said, in the Neotropical region where *Rhizophora mangle* predominates, it is easier to restore specific stands using this species due to its properties of resistance, ease of planting and rapid growth.

Worldwide, the use of pioneer or key tree species with more attributes like resistance, ease of planting and management, rapid growth, faster soil amelioration, high productivity and/or bearing some traditional use or fitting local environmental conditions, is the most common. For example, arid condition-resistant species such as *Avicennia germinans* (ACEP) or *Avicennia marina* (IWP) are indicated and used for restoration under such conditions. The Rhizophoraceae, including the genera *Rhizophora, Bruguiera* and *Ceriops*, are used most often because they have several attributes, but can also fail when selected areas do not have appropriate conditions for propagule rooting and development (hydrology, soil organic content and rhizosphere-associated microbiota, among others). *Rhizophora* in the ACEP, and *Sonneratia*, *Lumnitzera* and *Avicennia* in the IWP, can be used for initial plantings, since these pioneer species are able to colonize the littoral zone where tidal inundation and wave influences are stronger (Ferreira et al., 2023). With the increasing threat of climate change, it is possible to mitigate its impacts in coastal areas (storms, waves and extreme vents), using trees with more mechanical resistance and regeneration capacity after severe weather events. This needs to be carefully assessed, since root systems and trunks of *Rhizophora*, for example, can offer more resistance to storms and waves, but mature forests have no meristems to resprout after an extreme event, as does *Avicennia* (Villamayor et al., 2016).

After planting, procedures are the same as those used after natural or hydrologically-assisted recovery: monitor and assess the rehabilitation process examining environmental factors and processes, and the recovery of the biotic components (plants, fauna and microbiota). Several mangrove environmental services (e.g. clean water, high productivity, fisheries, etc.) aimed at recovery through forest restoration, depend directly or indirectly on the micro- and macro-biota, which in turn, also depend on these environmental services.

Mangrove ecosystem functionality is linked to functional groups of organisms that develop pivotal ecological roles or functions. Some key functional groups such as biogeochemical mediators (soil microbiota), bioturbators/burrowers, herbivores/detritivores, pollinators and wood borers are involved in processes that determine structural features, hence they are important in maintaining forest resistance and resilience, two crucial properties needed to face the increasing pressure of climate change drivers, such as storminess, sea level rise, the strength of ocean forcing and flooding (Ferreira et al., 2023, 2024). Crabs have been shown to be significantly resilient in recovered areas (Macintosh et al., 2002; Bosire et al., 2004). In the Philippines, replanted mangroves even in mono-genus stands (*Rhizophora* species) were effective in restoring mud crab (e.g. *Scylla olivacea)* populations, indicating recovery of their ecological function to a level equivalent to that of natural mangrove environments (Walton et al., 2007).

2.3. Examples from ACEP

There are many RRPAs, ranging from small (a few hectares), to intermediate, to extensive areas of several hundred hectares (Field, 1998; Primavera and Esteban, 2008; López-Portillo et al., 2017) mainly in S and SE Asian countries (Sri Lanka, Philippines, India and others), that have high tree diversity even at the single forest stand level. The problem is that, in general, large areas are restored with only one or a few tree species, usually for logistical reasons, which often results in functionally poor mangrove forests and even unsuccessful outcomes such as propagule dieback. The ACEP poses a completely different scenario.

RRPAs performed in mangroves in northeast Brazil (the region with the lowest percentage of mangrove coverage of ≈4%) and under a semi-arid climate, making mangrove conservation and functional recovery extremely important and are exemplary for further discussion. In the Potengi River estuary, in Rio Grande do Norte State, for example, in abandoned areas cleared for shrimp farming, a restoration program was initiated to restore the mangrove for the return of extractive fisheries (crustaceans, mollusks and fishes), crucial for the native communities' livelihood. In addition, the RRPA served as a template for studying the functional recovering capacity of areas impacted by aquaculture. Of two nearby cleared areas of mangrove, one was planted with the original species (*Rhizophora mangle*) and the other left to self-recover. After a few years, a relatively dense forest grew in both areas. The increase in biomass was higher in the planted area but remained monospecific due to high consumption of endemic incoming propagules of *Lumnitzera racemosa* (white mangrove) and *Avicennia germinans* (black mangrove), preferred by crabs. The self-recovered area took more time to recover and to reach the high biomass of the former but it recovered, colonized with the three most common local species, *Lumnitzera racemosa*, *Avicennia germinans* and *Rhizophora mangle* (red mangrove), due to low herbivorous crab populations. However, in this self-recovered area, red mangrove remains predominate over the other species after 16 years of management. Results showed that: i) the active and passive recovery of mangroves can exhibit similar outcomes in terms of biomass per hectare; ii) the predominance of higher biomass *Rhizophora mangle* trees and thus higher carbon stock in forest biomass (a common feature of neotropical mangroves), may be favored by higher consumption of white and black mangroves propagules by crabs; and iii) some significant faunal functional groups, such as burrower/bioturbators and herbivore/ omnivores, mostly consisting of Brachyuran crabs, show high resilience and can soon recolonize cleared areas, being associated with the predominance of *Rhizophora mangle* and consequently the higher primary productivity of that forest (Ferreira et al., 2015, 2019).

At the north of this site, in Ceará State (a semi-arid region with lower average annual rainfall than Rio Grande do Norte), an estuarine area of around three hectares of abandoned saltwork (formerly a salt flat, called *'apicuns'* in Brazil), was studied with the aim to recover mangroves in a Conservation Unit and avoid occupation by humans and/or cattle. The area was rehabilitated *via* different methods including hydrological restoration, planting of key tree species, and passive restoration with propagule settlement facilitated by nurse herbaceous species (*Sesuvium portulacastrum*, *Batis maritima* and *Eleocharis* species). The most resistant species to hypersaline soils, *Avicennia germinans,* and in lower density, *Lumnitzera racemosa,* were the major colonizers mostly inside herbaceous patches (Ferreira et al., 2022; Villavicencio et al., 2024). Because of functional groups (e.g. biogeochemical mediators, bioturbator crabs, herbivore crabs and insects), ecological interactions such as facilitation, herbivory and bioturbation increasingly shaped an establishing forest (Ferreira et al., 2022; Villavicencio et al., 2024). This confirms that salt flats are areas prone to colonization by mangroves when ongoing sea level rise pushes mangroves landward in Brazilian semi-arid estuaries (Godoy and Lacerda, 2015) and highlights the need to manage these salt flat ecosystems to avoid depletion of endemic species (e.g. the endangered great blue crab, *Cardisoma gunahumi*) by mangrove encroaching. In general, however, other salt flats coexist with mangrove fragments in the estuary and serve as biodiversity reservoirs, similar to a permanently changing patched structure or 'mosaic' of different forest successional formations and species assemblages (Alongi, 2009). This structure of mangrove stands allows mosaic-restoration using the spectra between passive and diverse active restoration techniques at different levels of degraded areas in the same mangrove forest.

Also in Ceará State, a small area of 1.75 ha located at the mouth of the Cocó River was restored by planting 3- and 7-year-old *Rhizophora mangle* plant fragments. Recent studies compared these planted fragments to natural and degraded areas nearby and found a trend towards higher fine sediment and carbon accumulation as mangrove age, with the highest values being found in the mature mangrove patch. Higher carbon contents resulting from soil inputs from vegetation through root growth and exudates, increased microbial biomass and plant litter (Jimenez et al., 2021) and are typical of *Rhizophora* forests (Ferreira et al., 2019b). This confirms the effectiveness of mangrove R/R to restore soil properties, as well as confirming the efficacy of *Rhizophora mangle* as a key species for neotropical mangrove rehabilitation.

Despite the increasing number of mangrove RRPAs, these measures are of small magnitude compared to the areas that climate change drivers and extreme events can damage, for example mangrove stands in Australia (Duke et al, 2022), Mozambique (Machava-António et al., 2014), the Caribbean and SE Asia, that have been subject to massive diebacks due to the increasing frequency and strength of storms. Presently, RRPAs need to address the eventual effects of climate change on target areas, a condition that is ubiquitous to the Anthropocene. Only by understanding that mangroves are significant 'life support ecosystems' and reinforcing their conservation as a priority State policy, will it be possible to halt the accelerated losses of these forests, their biological diversity and the valuable goods and services they provide (Lacerda et al., 2019).

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Chapter 7

The value of mangrove forests: An economic perspective

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1. Introduction

Mangroves are among the most productive ecosystems worldwide, providing a vital habitat for many species and a variety of provisioning, regulating, cultural, and supporting services for coastal communities (Das, 2020; Malik et al., 2023). Local communities harvest firewood, construction materials, finfish, crustaceans and other goods that improve their livelihood. They also benefit from regulating services provided by mangroves such as protection against storms and control of flooding and erosion. Mangroves also provide recreational, tourism and other cultural services, while supporting ecosystems by preventing soil salinity, retaining sediment and nutrients and nursing fish fry, for example. At the global level, mangroves play a crucial role in mitigating climate change due to their high carbon sequestration capacity (Choudhary et al., 2024).

Unfortunately, many mangrove forests are currently at risk of collapse due to climate change and related effects, as well as anthropogenic interventions such as urban development, agriculture, aquaculture, tourism and economic activities (Barbier, 2016; Ferreira et al., 2022). It is estimated that the global mangrove area decreased from 18.8 million ha in 1980 to 14.7 million ha in 2020 (Malik et al., 2023; Zhang et al., 2024). The relative importance of mangroves is oftentimes overlooked because of the lack of monetary values of several mangrove ecosystem services that are not traded in a market setting (Vo et al., 2012). Consequently, due to the limited information available, decision-makers tend to favor urban projects and economic developments over the conservation and restoration of mangrove forests (Hernández-Blanco et al., 2021). Monetary values reflecting individuals' preferences for the use of mangrove areas may assist policymakers to make better-informed decisions about funding conservation efforts or approving urban and economic development projects that may destroy mangrove forests.

Economists have developed a variety of methods to estimate values for ecosystem services using market data, monetary data in surrogate markets, and hypothetical scenarios when market data is unavailable (see Champ et al., 2017). These methods have been extensively applied to estimate values for different ecosystem services provided by mangrove forests worldwide (see reviews by Gargaran et al., 2024; Himes-Cornell et al., 2018; Mehvar et al., 2018; Vo et al., 2012). However, the economic concept of valuing ecosystem services in monetary terms is often misunderstood and therefore misused or even rejected by different stakeholders (Segerson, 2017). While the literature has standardized the presentation of value estimates in terms of hectare per year to show the importance of mangrove areas, existing mangrove valuation studies differ from each other in the services valued, methodology applied, and stakeholders with standing (i.e. groups who would experience benefits or bear the costs of the intervention) in the analysis. The differences may lead to misinterpretation of values estimated for particular mangrove ecosystem services as the total value of mangrove forests, thus underestimating the total value of those ecosystems.

This chapter aims to facilitate an appropriate usage of value estimates provided for mangrove ecosystem services and to provide guidance on how to estimate the total value of a mangrove forest. The rest of the chapter is organized as follows: Section 2 presents the concept of economic valuation of ecosystem services and describes some methods commonly used to estimate values for the services. Section 3 describes the literature on economic values of mangrove ecosystem services. Section 4 provides a roadmap to estimate the total economic value of a mangrove forest area. This may be particularly important for cost-benefit analyses of mangrove conservation projects. Section 5 concludes the chapter.

2. Conceptual framework and methods to value ecosystem services

From an economic perspective, the value of an ecosystem service is depicted by the maximum amount of something else that a person is willing to give up for that service. This implies that different individuals can measure the value of an ecosystem service in terms of different goods, services, or resources. Given the complexity of aggregating values measured in different units, economists have proposed to use money as a standard unit of measure. Such standardization allows us to estimate the value of a given ecosystem service as the maximum amount of money that an individual would pay for obtaining that service (i.e., willingness to pay), or the minimum monetary compensation that an individual would accept to give it up (i.e., willingness to accept) (Segerson, 2017). Nevertheless, other measure units can be used for estimating the value of ecosystem services. For instance, de Rezende et al. (2015) estimated values for mangrove restoration projects in Brazil in terms of contributions of both money and labor. Moreover, they argue that labor contributions can be a more practical measure of value for mangrove ecosystem services in areas with high levels of income poverty. This is certainly a matter that deserves more attention in the literature.

By definition, the economic valuation of ecosystem services is anthropogenic, reflecting individual preferences over different outcomes and depending on personal endowments that affect the individual capacity to make trade-offs (Segerson, 2017). The dependency on personal endowments is particularly important to understand because it may lead to differentials in economic values between affluent and poor individuals, even if they derive identical benefits from the ecosystem service. This is not an issue specific to monetary values. On the contrary, it may affect any measures of economic values including time contributions. Consequently, the standard practice of aggregating individual values to reflect the social preference for a given ecosystem service does not actually give equal weight to preferences of all individuals (Segerson, 2017). Of course, preferences can also be heterogenous across individuals. However, that heterogeneity is depicted by different types of values assigned to ecosystems services.

2.1. Typology of economic values

Figure 1 illustrates the intersection of standard classifications of economic values (in rectangles) and mangrove ecosystem services (in ovals). The total economic value (TEV) can be computed by aggregating use and non-use values across ecosystem services. Use values depict benefits that an individual derives from using ecosystem services, either directly or indirectly. For example, local communities can directly benefit from food services (e.g., fish and crustaceans), or indirectly from protection against storms provided by mangroves. On the other hand, individuals who do not currently use those services may still be willing to pay to maintain the ecosystem so that they can possibly use its services in the future (i.e., option value). It may also be the case that they have strong preferences for preserving the ecosystem so others can use its services now (i.e., altruistic value) or in the future (i.e., bequest value). Additionally, individuals may value the existence of an ecosystem even if they do not expect it to be used by anyone.

Mangrove ecosystem services may be classified within the TEV framework depending on their uses. For instance, individuals make direct use of provision and cultural services, while using regulating services only indirectly. This distinction is important because it helps researchers select an appropriate economic valuation method to estimate direct use values for provision and cultural services and indirect use values for regulating services. Figure 1 also shows that individuals may assign non-use values to provision, regulating, and cultural services. The estimation of non-use values requires a different type of valuation methods described in the next sub-section.[1] It is worth noting that supporting services are not included, not because they are not valuable, but because their value is considered in values of other services (Das, 2020).

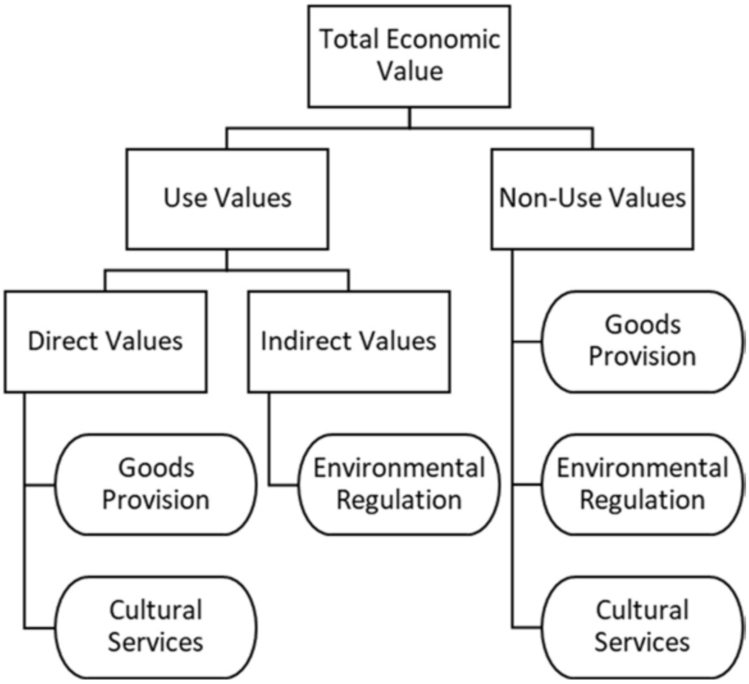

Figure 1. A typology of economic values applied to mangrove ecosystem services (adapted from Segerson, 2017 and Das, 2020).

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[1] Alternative classifications can be found in the literature. For instance, Das (2020) divided non-use values in existence and option values and argued that, while individuals assign both types of non-use values to cultural services, they only assign option values to provision and regulating services. In contrast, here I assume that individuals can assign other non-values such as altruistic and bequest values to provision and regulating services. This minor discrepancy regarding motives behind non-use values does not affect the availability and selection of methods suitable to estimate non-use values.

2.2. Economic valuation methods

Economists have developed multiple methods to analyze, and even generate, data on individual behaviors that reflect the value of a given ecosystem service. These methods can be classified depending on whether the usage of a given ecosystem service is actually observed in market settings, either directly or in surrogate markets, or stated in hypothetical scenarios (see Table 1).

Revealed preference methods rely on data obtained from observed usage of ecosystem services (e.g., timber harvesting and site visits) and can be suitable for estimating use values. However, these methods cannot depict non-use values which, in many contexts, may represent a substantial share of the total economic value. In contrast, stated preference methods can estimate both use and non-use values because they ask individuals to make choices in hypothetical scenarios that can include goods and services not traded in market settings. Some economists have raised concerns regarding stated preference methods, arguing that individuals may behave differently in a hypothetical setting where they do not face actual consequences for their stated choices. In response to that criticism, environmental and ecological economists have developed an extensive toolkit to mitigate potential biases introduced by the hypothetical nature of stated preference methods (see Haab et al., 2020; Holmes et al., 2017; Johnston et al., 2017; Mariel et al., 2021).

Table 1. A typology of selected valuation methods.

Methods	Revealed Preferences	Stated Preferences
Direct	• Market price • Replacement cost • Avoided damage	• Contingent valuation
Indirect	• Travel cost • Hedonics	• Discrete choice experiments

2.2.1. Market prices

When ecosystem goods are traded in a well-functioning market, market prices can be used to value that good. For instance, market prices can be used to estimate the value of catch by comparing prices of fish from areas with and without mangroves (Gargaran et al., 2024). Market prices are usually used with other revealed preference methods (e.g., hedonic pricing and replacement cost approaches). Although market prices are among the most commonly used approaches to value mangrove ecosystem services (Himes-Cornell et al., 2018), it is also true that several services are not traded in market settings and, consequently, require other methods to be valued.

2.2.2. Replacement cost

This method relies on the notion that the value of a mangrove ecosystem can be estimated by calculating the cost of replacing it with man-made infrastructure to deliver the same service, e.g., the cost of building a seawall that replicates the protective services provided by the mangrove forest. See Brown (2017) for a detailed discussion of this method.

2.2.3. Avoided damage

This method is suitable to estimate the value of regulating services provided by mangrove ecosystems (e.g., flood and erosion control). The method assumes that the value of these services is equivalent to the value of economic activity, property and even human lives that would be lost in the absence of the mangrove ecosystem (Barbier, 2016).

2.2.4. Travel cost

The value of recreational services (e.g., tourism and bird-watching) can be estimated by aggregating the travel expenses, including the opportunity cost of travel time, incurred to visit the site. Like any other approach, the travel cost method has some limitations such as the lack of consensus on an appropriate cost of travel time. Additionally, this method is not suitable to value goods that are not consumed *in situ* (Birol et al., 2008).

2.2.5. Hedonic pricing model

The premise underlying this approach is that goods are bundles of amenities and, consequently, the market price of that good is a collection of values assigned to each of the amenities. This approach is usually applied to housing prices, under the assumption that the value of ecosystem services is capitalized in property values. Similar to the Travel Cost method, the Hedonic Pricing Model can only value goods consumed *in situ* (Birol et al., 2008).

2.2.6. Contingent valuation method

This method is suitable to elicit preferences for changes in ecosystem services in monetary terms. It is based on surveys that ask respondents about their maximum willingness to pay or minimum willingness to accept, depending on property rights, for a hypothetical change in a given ecosystem service. Economists have developed a set of guidelines and tools aimed to minimize hypothetical biases in values estimated using this approach (see Haab et al., 2020; Johnston et al., 2017).

2.2.7. Discrete choice experiments

This survey-based approach consists of hypothetical scenarios where respondents are asked to choose among two or more alternatives with different attributes of ecosystem services. By choosing one of the alternatives, respondents state their preferences regarding specific attributes. Moreover, the inclusion of a payment that can vary across alternatives allows for estimating respondents' willingness to pay for different levels of ecosystem services (Holmes et al., 2017). Despite its hypothetical nature, discrete choice experiments have proven to be effective in eliciting preferences for ecosystem services that are not traded in a market.

2.2.8. Benefit transfer function

Economic valuation may require a substantial amount of time and financial resources. When these resources are not available, the benefit transfer approach may be an alternative. In this method, existing value estimates from different sites are used to estimate values in the site of interest. In a recent review of the related literature, Himes-Cornell et al. (2018) found that 32 out of 70 studies conducted between 2007 and 2016 used benefit transfer to estimate the value of mangrove ecosystem services.

3. Valuation studies on mangrove ecosystem services

Many valuation studies on mangrove ecosystem services have been conducted worldwide (Mehvar et al., 2018), usually presenting value estimates per hectare per year. The majority of these studies have taken place in South and Southeast Asia. Other regions with less coverage in the literature include Oceania, Africa, and Central and South America (Himes-Cornell et al., 2018).

Existing studies vary in terms of the ecosystem services valued and valuation method utilized. Provision services such as food and raw materials have received the most attention in the literature, followed by regulating services such as climate regulation and protection against extreme weather events. In contrast, cultural and supporting services have been valued in fewer studies (Gargaran et al., 2024; Getzner and Islam, 2020; Himes-Cornell et al., 2018). Given that several provision services are traded in market settings, it is not surprising that market prices have been the most popular approach to estimate the value of mangrove forests (Gargaran et al., 2024). The benefit transfer method has also been used by a large number of studies (Himes-Cornell et al. 2018), presumably due to the convenience of using existing values estimated in other sites. The replacement cost and avoided damage methods are applied to value regulating services (Barbier, 2016; Gargaran et al., 2024). Whereas stated preference methods are mostly used to value cultural services (Gargaran et al., 2024).

The average values of ecosystem mangrove services reported in recent meta-analyses tend to differ from each other, leading to different conclusions about which mangrove ecosystem services are more valuable. As an example, Table 2 compares average values for different mangrove ecosystem services reported by Getzner and Islam (2020) and Gargaran et al., (2024). Based on 250 values from 66 primary valuation studies conducted worldwide, Getzner and Islam (2020) reported the average values suggesting that cultural services are the most valuable, followed by regulating services. In contrast, Gargaran et al., (2024) analyzed 194 value estimates from 12 Asian countries. Their respective average values suggest that provision and regulating services are more valuable than supporting and cultural services.

Table 2. Average values of mangrove ecosystem services in recent meta-analysis (in 2020 US$ ha^{-1} $year^{-1}$).

	Getzner and Islam (2020)	Gargaran et al. (2024)
Provisioning services	5,048	7,258
Regulating services	37,208	6,580
Supporting services	414	1,350
Cultural services	50,811	1,372

It could be argued that Getzner and Islam (2020) and Gargaran et al., (2024) reached different conclusions regarding the value of mangrove ecosystems services due to the geographical scope of their review. Yet, the range of estimated values analyzed by each of these meta-analyses is substantially wide too. Getzner and Islam (2020) concluded that the variation in estimated values can be partially explained by differences in valuation methods applied, the types of ecosystem services considered, and the conservation status of the mangrove forest. However, differences explained less than 25% of the variation in estimated values, suggesting that there are unobserved factors (e.g., local preferences) underlying the wide range of estimated values.

Himes-Cornell et al. (2018) pointed to a potential overuse of the benefit transfer method. They showed substantial gaps between the average value of different services estimated using a benefit transfer approach and the average value of corresponding services obtained using other methodologies. For instance, they reported an average value of US$ 24,312 ha^{-1} $year^{-1}$ for food services if benefit transfer studies are excluded vs. US$ 3,609 ha^{-1} $year^{-1}$ if only benefit transfer studies are considered. This gap may reflect differentials in preferences and prices between study sites. Similarly, in a comparison of benefit transfer values and values estimated using primary valuation methods conducted in Costa Rica, Hernández-Blanco et al., (2021) found that the benefit transfer method may overestimate the TEV of a mangrove area. However, they found that benefit transfer methods may yield similar values if local expert knowledge is elicited to identify ecosystem services used by local populations and mangrove areas that provide the services.

Existing value estimates should be used with caution for making decisions regarding conservation of mangrove areas. Many studies provide use values for specific mangrove ecosystem services (Gargaran et al., 2024; Getzner and Islam. 2020). However, fewer studies have estimated non-use values for mangrove ecosystem services (Himes-Cornell et al., 2018). Moreover, some mangrove ecosystem services have not been considered in the valuation literature. Consequently, it is difficult to compute the TEV of a given mangrove area using estimates from the current literature.

4. A roadmap to estimate total economic values of mangroves

Estimating the TEV of mangrove forests is challenging given that each stakeholder benefits from multiple mangrove ecosystem services in a different way, including nonusers who may assign a value to the existence of mangrove forests. Hence, economic valuation of mangroves requires careful design, resources and expertise from different disciplines (Das, 2020).

Table 3 proposes a multi-stage process to conduct valuation studies on mangrove ecosystem services. The initial step is to identify potential alternatives for the mangrove area to be analyzed. Examples of salient interventions include mangrove restoration programs, urban development, and land conversion to aquaculture and agriculture, among others. This information is usually found in official documents of environmental and urban planning agencies. Gray literature can also be a source of information, although it should be triangulated using qualitative interviews with local authorities. In addition to learning about specific plans, projects and programs that may affect the mangrove forest, this is a great opportunity to engage decision-makers as they will ultimately be the users of value estimates.

Table 3. Valuation process, adapted from Segerson (2017).

Step 1	Problem Characterization
Step 2	Identifying significant ecosystem changes
Step 3	Identifying the impacts of ecosystem changes on ecosystem services
Step 4	Predicting the magnitude of changes in ecosystem services
Step 5	Estimating economic values
Step 6	Communicating the results to relevant decision-makers

Interventions identified in the first stage may impact the forest area, the condition of its vegetation, and other biophysical characteristics of the forest. Hence, identifying corresponding changes in mangrove ecosystems requires an interdisciplinary approach. The need for an interdisciplinary team is even more obvious in Step 3 of the valuation process. In that stage, the valuation team identifies the provision, regulation and cultural services that could be impacted as a result of changes in the mangrove ecosystem. Ecologists, biologists, chemists, hydrologists and other natural scientists play an important role in predicting changes in different characteristics of mangrove forests and respective ecosystem services. The valuation team also identifies groups with standing and the mangrove ecosystem services that are relevant for them, which combined with the findings of natural scientists, allows them to paint a more complete picture of potential consequences of the interventions under evaluation. Social scientists may assist in this task.

Next, changes in mangrove ecosystem services need to be quantified. Those measures must be expressed in terms that groups with standing can easily understand and value. For instance, local populations may not understand scientific measures of microphytobenthos in an additional hectare of mangrove forest. Instead, they may more easily interpret an increase in the population of crustaceans sustained by the extended mangrove area, especially if those crustaceans are part of their diet and livelihood.

With a clear definition and measures of the mangrove ecosystem services to be valued, the valuation team selects the appropriate methods. Several factors play a role in this decision including information, time and financial constraints. The use of primary valuation methods (see Section 2) is highly recommended over transferring values from previous studies conducted in areas that can be quite different than the study site (Getzner and Islam, 2020; Himes-Cornell et al., 2018). However, when data, resources, or expertise are not available, the benefit transfer method with proper adjustments is a feasible option (Das, 2020; Hernández-Blanco et al., 2021).

Finally, findings should be shared with decision-makers and other stakeholders using effective language and proper channels (e.g., presentations, reports, policy briefs and scholarly publications). Value estimates should be interpreted for different audiences, along with their policy implications within the local context. The scope and limitations of the valuation study should also be reported.

5. Conclusions

The importance of mangrove ecosystems has motivated an extensive number of valuation studies synthetized in recent meta-analyses (e.g., Gargaran et al., 2024; Getzner and Islam, 2020; Himes-Cornell et al., 2018; Mehvar et al., 2018). Those studies have applied a variety of valuation methods to estimate values for a litany of provision and regulating services. In contrast, few studies have estimated values for cultural services that can be equally (or more) important for local populations (Getzner and Islam, 2020; Himes-Cornell et al., 2018). Another shortcoming in the literature is the estimation of non-use values. These limitations make it difficult to estimate TEV of mangrove forests, which may lead decision-makers to favor other land uses over preserving mangrove areas. Logical extensions to the literature include the valuation of cultural services and the estimation of non-use values for all types of mangrove ecosystem services.

Transferring values from existing studies could be an alternative to estimate the TEV of mangrove forests if these values were estimated in similar sites. However, most valuation studies have been conducted in Asia (Himes-Cornell et al., 2018), making it difficult to transfer values to other regions where populations may have different preferences and therefore may assign different values to similar ecosystem services. More primary valuation studies on mangrove ecosystem services are needed, particularly in regions other than Asia, to facilitate the transfer of values across similar sites.

In general, economic values lead to important decisions regarding the management and preservation of productive ecosystems such as mangroves. Yet, the valuation of mangrove ecosystems services has often been misunderstood as a neoclassical attempt to impose a price on public goods provided by these ecosystems. However, as Vo et al. (2012) argued, ‘society is governed by money and numbers, and if we do not put a value on ecosystem services, they might be ignored in favor of the quantifiable.’ The anthropogenic nature of the valuation approach is controversial as well, particularly among natural scientists who recognize the importance of mangrove areas for non-human species. There is room in the economic valuation approach for species provided individuals assign value to their existence. In fact, estimating non-use values for the species may help demonstrate the importance of mangroves beyond the usage of their ecosystem services.

Estimating values for mangrove ecosystem services is not without challenges. However, there is a set of well-developed methodologies suitable to undertake the task that is vitally important to the conservation of this ecosystem. This chapter described a multi-stage process that interdisciplinary teams can follow to estimate values for specific mangrove ecosystem services and, in turn, the total economic value of mangrove forests. It is my hope that this chapter will help in bringing economists and natural scientists together so they can show the relevance of mangrove ecosystems to decision-makers and other stakeholders.

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Chapter 8

Community-based management and co-management of mangroves in the Anthropocene: A global review of current topics and trends

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1. Introduction

Mangroves represent one of Earth's most remarkable ecosystems, serving as critical transition zones between terrestrial and marine environments in tropical and subtropical regions worldwide. These unique coastal forests are invaluable in supporting global biodiversity, protecting coastlines from erosion and storm damage, and providing essential socio-economic benefits to local communities. In recent years, their significance in climate change mitigation has gained increasing recognition due to their exceptional capacity for carbon sequestration and storage, earning them the designation of blue carbon ecosystems.

However, these vital ecosystems face unprecedented threats from multiple fronts. Unsustainable aquaculture practices, particularly the expansion of shrimp farming, continue to drive mangrove deforestation. Coastal development, industrial pollution, and the accelerating impacts of climate change further compound these pressures (Quinn et al., 2017). In response to these challenges, conservation approaches have evolved from traditional top-down management to more inclusive strategies that recognize the integral role of local communities in ecosystem stewardship (Crona and Hubacek, 2010).

Two approaches have emerged as particularly promising: Community-Based Management (CBM) and Co-Management (CM). While often discussed interchangeably, these approaches represent distinct paradigms in natural resource management. CBM emphasizes devolving rights, responsibilities and authority from central governments to local communities, building on traditional ecological knowledge and local governance systems. In contrast, CM represents a shared management approach, fostering partnerships among government agencies, local communities and other stakeholders.

Although CBM and CM emphasize different aspects of governance, CBM focusing on local community authority and CM highlighting shared decision-making these approaches are not necessarily mutually exclusive. In practice, they often overlap, as many co-management arrangements incorporate elements of community-driven governance, and community-based initiatives frequently involve collaboration with external actors. Given this conceptual proximity, in this review, we treat CBM and CM as interconnected frameworks rather than strictly distinct categories, recognizing that both emphasize participatory governance and the active role of local communities in resource management.

Despite indications of a growing adoption of participatory management strategies, a comprehensive, up-to-date synthesis of their effectiveness in mangrove conservation is lacking. To our knowledge, the last global review of CBM in mangrove conservation was conducted by Datta et al. (2012). Over the past decade, significant changes including shifts in governance structures, new financial mechanisms, and evolving socio-political contexts necessitate a reassessment of how CBM and CM contribute to mangrove conservation outcomes today. Datta et al. (2012) reviewed CBM and not CM, while we decided here to include CBM and CM precisely to obtain a more comprehensive review, given that the terms can be treated as synonyms; for both, the state or funding agencies appear as the main actors in terms of the initiative to implement management that considers the role of the community.

1.1. Research questions and objectives

Aiming to update and expand the work done by Datta et al. (2012), this chapter addresses two fundamental questions in the context of global mangrove conservation: 1). How do CBM and CM schemes contribute to mangrove conservation outcomes across different geographical and socio-cultural contexts? 2). What critical lessons can be drawn from existing implementations to inform and improve future conservation initiatives? Through these questions, we aim to briefly synthesize existing knowledge on CBM and CM initiatives worldwide.

2. Materials and methods

2.1. Systematic review protocol

We employed a state-of-the-art review approach following a protocol to ensure methodological rigor and reproducibility. This was supplemented by the SODIP (Search, Organize, Document, Interpret, Present) approach, which provided additional structure to our review process.

2.2. Search strategy and data collection

For our literature search, in English only, we utilized the database Web of Science (WoS). The search strategy employed a carefully constructed query combining key terms related to mangroves, management approaches and conservation outcomes. The core search query included: TI = mangrove* AND co-manag* OR co-manag* OR community-based conserv* OR community-based manag* OR collaborat* OR participat* OR local communit* OR community-based conserv* OR participatory conserv* OR CPR manag* OR common-pool resource manag* AND manag* OR conserv* OR protected area*.

This query was applied exclusively to the title and keyword fields to avoid retrieving papers that mention management only tangentially for instance, as a potential beneficiary of their findings or as an area requiring further research. From the initial set of papers identified, a subset was selected for in-depth qualitative analysis ($n = 39$), which is further explored in the Results and Discussion section. The selection process involved careful review by two authors of this manuscript (RB and IE), who independently assessed the abstracts to determine whether a publication was suitable for qualitative investigation. Only papers selected by both authors were included in the qualitative analysis.

2.3. Inclusion and exclusion criteria

We developed clear criteria to guide document selection:

Inclusion Criteria: Studies directly addressing CBM or CM of mangroves; case studies from tropical regions; and peer-reviewed articles and chapters.

Exclusion Criteria: Studies unrelated to mangrove ecosystems; research conducted outside tropical regions; and publications not addressing management approaches directly.

2.4. Data analysis

Our analysis combined quantitative and qualitative approaches. We used XLStat and Votant tools for statistical analysis, while VosViewer facilitated bibliometric analysis and visualization of research networks. Spatial distribution of studies was mapped using QGIS v3.16, incorporating data from GLOBIL and Natural Earth.

3. Results and discussion

3.1. Annual distribution, types of publications and geographical scope

The WoS search yielded 113 publications. The temporal analysis of publications reveals a significant shift in research attention towards community-based and co-management approaches in mangrove conservation (Figure 1). While publications remained relatively sparse through the early 2000s and early 2010s, with only two or three publications annually, a marked increase began around 2016-2017. This surge culminated in a notable peak of approximately 12 publications in 2018, followed by sustained higher publication rates in subsequent years. Research articles constitute the primary publication type throughout the period, demonstrating the field's emphasis on empirical studies. However, the emergence of review papers since 2020, with an earlier exception in 2012, suggests the field has matured sufficiently to warrant synthetic analyses. The presence of proceedings papers and meeting abstracts in recent years further indicates active academic discourse and knowledge exchange in the field. However, the annual publication output remains small for robust statistical analyses.

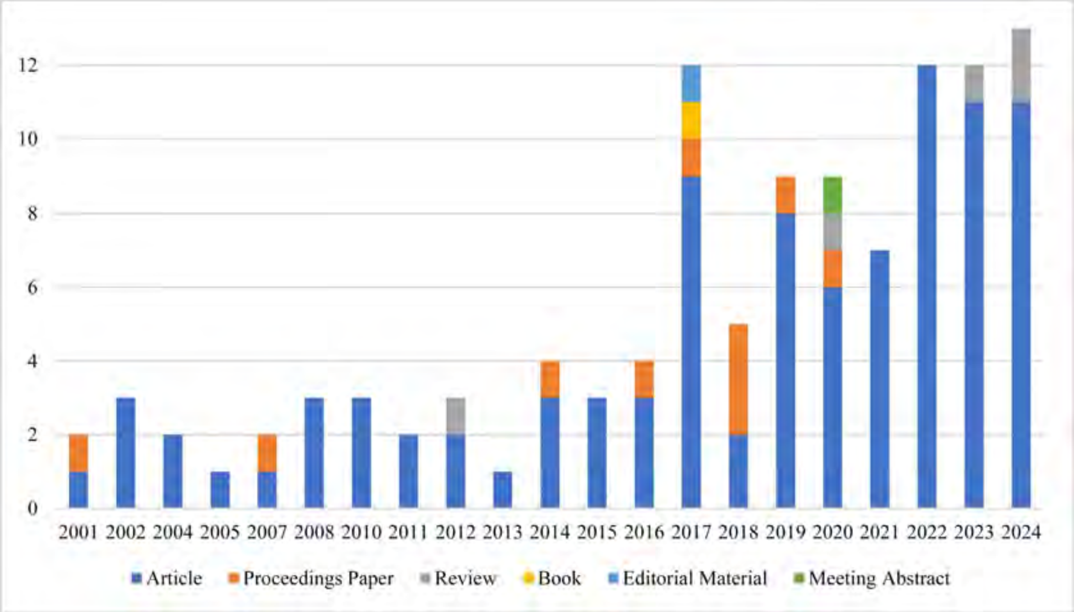

Figure 1. Annual number of publications (y-axis) found in the search with an indication of the type of publication. The temporal scope spanned from the 2000s to 2024, capturing the evolution of management approaches over two decades.

The geographical distribution of studies demonstrates a pronounced concentration in Southeast Asia (Figure 2). This regional focus aligns with the area's global significance in mangrove ecology, as it hosts approximately one-third of the world's mangrove forests (Spalding et al., 2010). The pie chart and distribution map reveal that while Southeast Asian case studies dominate the literature, other regions with significant mangrove ecosystems remain comparatively understudied. This geographical skew, while reflecting the importance of Southeast Asian mangroves, also highlights potential research gaps in other critical regions such as Africa, Latin America and Oceania, where CBM approaches might offer valuable conservation solutions.

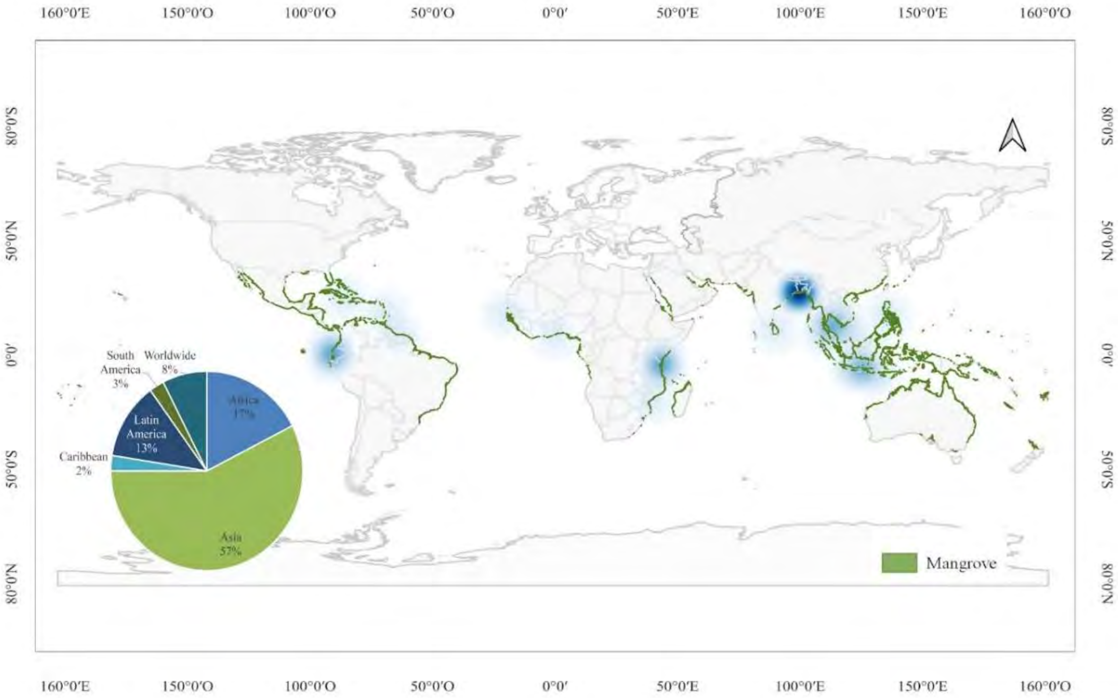

Figure 2. Geographical distribution of studies captured in the search, showing a clear hotspot in Southeast Asia.

The word clouds derived from different sections of the publications offer insights into the conceptual framework and evolution of the field (Figure 3). The most frequent words are, as expected, the words used in the search query. Overall, the three sections (title, keywords, abstract) do not differ much in terms of frequency of words. The title word cloud (Figure 3a) emphasizes the central role of mangroves and conservation, while highlighting the significance of 'community' and 'local' (as in 'local communities' or 'local knowledge'). The keyword analysis (Figure 3b) reveals 'community-based' and 'management' as the dominant terms. The keyword section also more evidently brings the aspect of participation. The abstract (Figure 3c) brings in elements such as 'local' like the titles and 'resources'.

A Sankey diagram visualization reveals the complex inter-connections between countries, continents and research topics in community-based mangrove management studies (Figure 4).

At the country level, Bangladesh, Indonesia and Thailand emerge as the most prominent research locations in Asia, while countries like Ghana, Tanzania and Mozambique represent key study areas in Africa. The diagram illustrates how research from these locations flows through continental regions (Asia, Africa, Latin America and the Caribbean) to connect with diverse research topics.

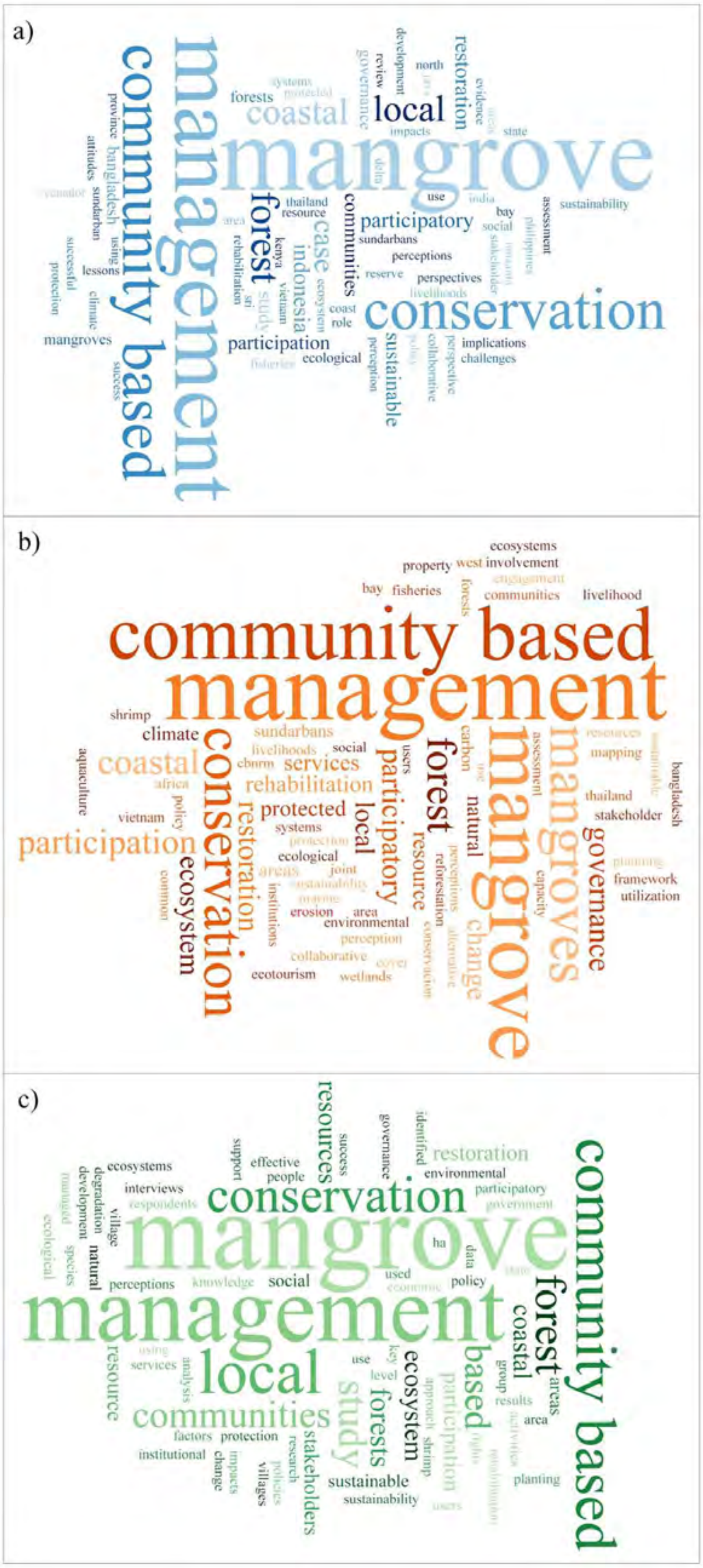

Figure 3. Word clouds generated from the: a) titles, b) keywords and c) abstracts of the publications.

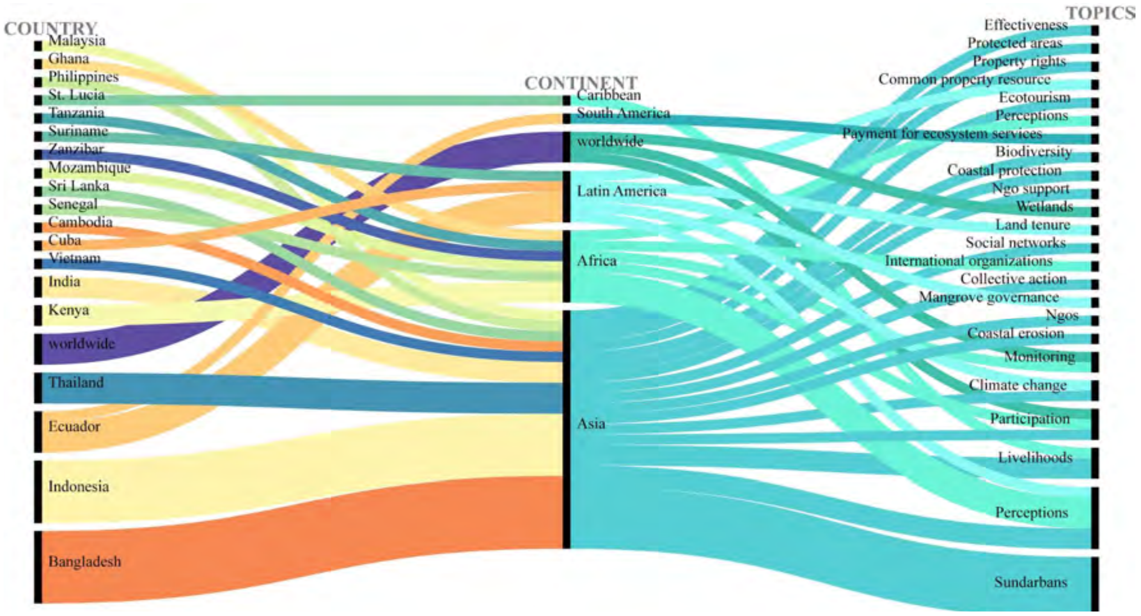

Figure 4. Study topics by country and continent.

3.2. Overview of topics approached in the publications

This review synthesizes diverse perspectives on the performance, equity considerations and socio-ecological impacts of mangrove co-management while also identifying a prominent trend in the literature: the call for increased collaboration among governments, NGOs and other external actors. In this short, state-of-the-art review, we aim to illuminate the opportunities, challenges and complexities inherent in mangrove co-management by analyzing key studies and case examples.

The topics addressed by the studies, as can be seen in Figure 4, reveal a comprehensive scope of mangrove management concerns, with protected areas, property rights and common property resource management featuring prominently. Notably, the diagram also captures emerging themes such as payment for ecosystem services, ecotourism and climate change. Social dimensions are well-represented through topics like perceptions, social networks and collective action, indicating the field's strong emphasis on human dimensions of conservation. The presence of topics such as NGO support and international organizations suggests the important role of external stakeholders in community-based initiatives.

In this short review, we also explore the multi-faceted dimensions of mangrove co-management, emphasizing its performance, equity considerations and socio-ecological impacts. Key topics include good governance in protected areas (Mollick et al., 2021), tenure and property rights (Roy and Gow, 2015; Roy, 2014), capacity building for management (Roy and Gow, 2015), and the relationship between community-based management and biodiversity conservation (Damastuti et al., 2022).

These studies confirm the intuitive assumption that mangrove co-management and community-based initiatives often are designed with the aim to balance biodiversity conservation with socio-economic benefits such as sustainable livelihoods (Datta et al., 2010; Mollick et al., 2021). However, these dual objectives can create tensions, even though it can be argued that these objectives complement each other and are not necessarily mutually exclusive. Still, these tensions can be found in mangrove co-management and are reported in the studies we found. For example, hierarchical committees—intended to enhance conservation—sometimes engage in destructive practices such as poisoning wetlands for fishing (Rahman, 2022). Mangrove co-management operates within complex socio-ecological systems, where interactions between human and natural systems often yield such tensions. This highlights the need for further investigations into trade-offs and connections between environmental conservation and the protection of the livelihoods of local populations.

3.2.1. Effectiveness and equity in mangrove co-management

A primary focus of co-management evaluations in the studies found in our search is its effectiveness for biodiversity conservation and local livelihoods (Datta et al., 2010; Damastuti and de Groot, 2017; Mollick et al., 2021; Damastuti et al., 2022, 2023). Studies undertaken by Datta et al. (2010) highlighted the potential for CM to deliver positive ecological outcomes, such as reduced deforestation, while fostering increased access to resources for local stakeholders. Oftentimes, effectiveness assessments fail to capture the complex realities of conservation governance, particularly the neoliberal tendencies that prioritize cost-effectiveness over equity and inclusivity. Neoliberal conservation often reorients human-environment relationships along capitalist lines, emphasizing market-based solutions such as payments for ecosystem services, ecotourism and carbon offset schemes while sidelining broader socio-political concerns (Holmes and Cavanaugh, 2012). These approaches tend to favor economic efficiency and measurable ecological outcomes, yet they frequently overlook the social inequalities embedded in conservation programs (Pascual et al., 2014). Such management approaches are largely based on perceptions (Jumnongsong et al., 2015; DasGupta and Shaw, 2017; Nop et al., 2017; Idrus et al., 2019; Owuor et al., 2019; Gallup et al., 2020; Martínez-Espinosa et al., 2020; Gayo, 2022; Ntibona et al., 2022; Begum et al., 2023; Djosetro and Behagel, 2024; Macamo et al., 2024; Portorreal et al., 2024). These perceptions and their analyses can help address the need to take into consideration social-ecological connections and justice issues, moving away from mere biodiversity conservation or socio-economic indicators.

Additionally, the growing reliance on non-state actors, such as private organizations and NGOs, in conservation governance, was reported by some studies analyzed here (Kongkeaw et al., 2019; Miller et al., 2019; Gallup et al., 2020). This reliance can hamper accountability, as decision-making authority shifts away from local communities to external stakeholders with financial leverage (Holmes and Cavanaugh, 2012). These dynamics illustrate the limitations of neoliberal conservation strategies and underscore the need for frameworks that integrate equity, participation and social justice alongside ecological objectives.

In line with the need to incorporate more diverse perspectives on success and effectiveness, studies show that stakeholder perceptions provide valuable insights into the equity dimensions of co-management. Begum et al. (2023), for example, reported that, while participants appreciate enhanced knowledge and resource access, they face challenges such as limited harvesting opportunities, insufficient support during decision-making processes, and restrictive membership rules. This same study shows that gender inequities are particularly pervasive in the Sundarbans mangrove forest, where women's contributions to conservation are systematically under-valued.

Funding constraints are a recurring challenge in mangrove co-management. Nop et al. (2017) and Pulhin et al. (2017) highlighted the need for stronger financial support to sustain community-based initiatives, while Nijamdeen et al. (2023) emphasized the importance of long-term funding for effective mangrove management. Damastuti et al. (2022, 2023) demonstrated that CBM initiatives can lead to successful biodiversity conservation and coastal protection when supported by long-term funding and technical assistance. Similarly, Gallup et al. (2020) noted that financial incentives from international organizations and NGOs play a significant role in motivating community participation in mangrove reforestation.

The inequities among stakeholders shown by the studies reflect broader structural issues within co-management systems, where marginalized groups often have limited influence over governance decisions. DasGupta and Shaw (2017) identified trust deficits between communities and officials as a key barrier to effective participation. Ntibona et al. (2022) also emphasized the need for inclusive governance structures that ensure equitable participation and benefit-sharing among stakeholders. Without this, marginalized groups, such as women and subsistence users, are often excluded from decision-making processes. Addressing these inequities requires efforts to create inclusive decision-making frameworks that amplify the voices of under-represented stakeholders.

Similarly, Jumnongsong et al. (2015) highlighted the complexities of aligning community perceptions with ecological realities, demonstrating that fishers' perceptions of mangrove benefits and threats often diverge from field-assessed conditions. This underscores the importance of participatory initiatives that foster trust, collaboration and shared responsibility among stakeholders, promoting a dialog of epistemologies and a platform for mutual understanding where conflicting views can be discussed and debated.

Broader structural issues within co-management systems, such as unequal power dynamics and limited support for alternative livelihoods, also hinder equity outcomes. Sukuryadi et al. (2020) in Indonesia and Aheto et al. (2016) in Ghana stressed on the importance of alternative livelihoods and income-generating activities to reduce dependency on mangrove resources.

De Oliveira et al. (2024) identified 61 governance drivers affecting the governance of coastal wetlands, including strong networks, government support and most importantly, local community participation. Their findings emphasize the need for decentralized and multilevel governance approaches that provide marginalized groups with equitable opportunities to engage in decision-making.

Incorporating local knowledge and practices into management strategies enhances conservation outcomes. Studies by Djosetro and Behagel (2024) in Suriname and Portorreal et al. (2024) in Cuba highlighted the value of local knowledge in designing effective conservation plans.

Several studies call for stronger participation or collaboration with state or government actors. For example, Datta et al. (2010) and Mollick et al. (2021) emphasized the need for state support in policy implementation and governance, while Damastuti et al. (2023) highlighted the role of government agencies in providing technical and financial assistance. On the other hand, studies by Sudtongkong and Webb (2008) and Miller et al. (2019) advocated for greater community autonomy, demonstrating that community-led approaches can achieve both ecological and socio-economic outcomes. Beitl (2011) and Roy and Gow (2015) further argued that weaker state involvement and expanded community control are key to sustainable mangrove management.

3.2.2. Regional patterns and success factors

The implementation of mangrove CM worldwide exhibits distinct regional patterns. Southeast Asia, particularly Indonesia, the Philippines and Thailand, is a hotspot for these initiatives and has emerged as a hotspot for community and CM practices, driven by strong government and NGO involvement (Kongkeaw et al., 2019). In contrast, Datta and colleagues found fewer initiatives in South America and Africa in 2012, though significant examples existed back then in Brazil, Tanzania, Madagascar and Kenya (Datta et al., 2012). Our current review, we believe, was not able to capture a faithful picture of the studies on CM and community-based initiatives in Latin America and the Caribbean. This is a limitation of the present study, which can be overcome by a more comprehensive review, which encompasses publications in other languages, especially Portuguese and Spanish.

In Indonesia, Damastuti and colleagues examined co-management's contributions to biodiversity conservation (Damastuti et al., 2022), coastal protection (Damastuti et al., 2023) and sustainable resource use (Damastuti and de Groot, 2017). Across all aspects, these authors found that success was linked to complementary governance measures, capacity building and inclusive participation. Recommendations include increased collaboration with governments and external groups to shift toward co-management arrangements with broader governance perspectives. These studies emphasized the importance of adaptive management practices and participatory ecosystem service mapping in achieving these goals.

The Sundarbans, spanning India and Bangladesh, is a focal point for studies on mangrove conservation, particularly through CBM and CM. Datta et al. (2010) highlighted the success of eco-development committees in the Indian Sundarbans, demonstrating a positive correlation between community participation and mangrove condition, while emphasizing the need for robust legal and institutional frameworks. In Bangladesh, Mollick et al. (2021) evaluated co-management in the Sundarbans, noting improvements in governance principles such as transparency and participation but identifying challenges in equitable benefit distribution and policy implementation. Similarly, DasGupta and Shaw (2017) examined incentive design in the Indian Sundarbans, revealing that restrictive state policies and trust deficits hinder community participation, calling for more inclusive and community-centered approaches. Begum et al. (2023) explored stakeholder perceptions in the Sundarbans, finding mixed views on co-management and underscoring the need for greater inclusivity, particularly for women and marginalized groups. Siddique et al. (2024) further assessed CM in the Sundarbans, highlighting its potential to improve conservation outcomes but noting persistent issues with equity, access and institutional support. Together, these studies underscored the Sundarbans as a critical site for understanding the complexities of mangrove conservation, where balancing ecological goals with socio-economic needs and fostering inclusive governance remain key challenges.

3.2.3. Challenges and future directions

Key challenges in CM include financial constraints, capacity limitations and institutional weaknesses. DasGupta and Shaw (2017) identified barriers such as restricted access to mangrove products, passive involvement of vulnerable groups, and trust deficits between communities and officials. Addressing these challenges requires innovative incentives, capacity-building initiatives and stronger partnerships among stakeholders. De Oliveira et al. (2024) advocated for governance reforms that address structural barriers such as unequal power dynamics, and promote inclusivity at all levels. Future research should investigate institutional and socio-political dimensions of co-management, including the impacts of globalization on local communities. Strengthening collaborations among governments, NGOs and local stakeholders will be critical to ensuring long-term success.

As previously discussed, a more in-depth investigation is needed into the different conceptualizations of effectiveness and success of mangrove social-ecological systems and the implications of a clear-cut separation between biodiversity conservation and the maintenance of local livelihoods.

3.2.4. Summary of information and publications

A summary of information and publications used in this study is shown in Table 1. The information is from six areas (Africa, Asia, Latin America, North America, South America and Worldwide). Topics included perceptions, international organizations, monitoring, participation, livelihoods, climate change, coastal erosion, NGOs, property rights, social equity, collective actions, social networks, biodiversity, protected areas, participatory management, co-management, coastal protection, effectiveness, mangrove communities, ecotourism, common property resource, land tenure, mangrove governance, payment for ecosystem services and governance of wetlands. The information was sourced from 40 references.

Table 1. Summary of information and publications used in this study

Area	Country	Topic	Type	Title	Author keyword	Research area	Reference
Africa	Zanzibar	Perceptions	Article	Local community perception on the State Governance of mangroves in Western Indian coast of Kinondoni and Bagamoyo, Tanzania	Mangroves; Local community perception; State management; Community; Involvement; Local livelihood	Biodiversity & Conservation	Gayo (2022)
	Mozambique	Perceptions		Mangrove community-based management in Eastern Africa: Experiences from rural Mozambique	Mangrove restoration; Law enforcement; Management of commons; Natural resources sustainable management; Community engagement	Environmental Sciences & Ecology	Macamo et al. (2024)
	Tanzania	Perceptions		Incentives and disincentives of mangrove conservation on local livelihoods in the Rufiji Delta, Tanzania	Mangroves; Conservation measures; Local communities; Livelihoods; Rufiji Delta	Forestry	Ntibona et al. (2022)
	Senegal	International organizations		Mangrove use and management within the Sine-Saloum Delta, Senegal	Coastal management; Community-based natural resource management; Fuelwood harvesting; Mangrove reforestation; West Africa	Oceanography	Gallup et al. (2020)
	Ghana	Monitoring		Community-based mangrove management, implications for local livelihoods and coastal resource conservation along the Volta estuary catchment area of Ghana	Community-based mangrove forestry; Mangrove restoration; Resource conservation; Livelihoods; Ghana		Aheto et al. (2016)
	Kenya	Participation		Involvement, knowledge, and perception in a nature reserve under participatory management: Mida Creek, Kenya	Mangrove; Participatory-forest management; Decentralization; Kenya		Frank et al. (2017)
	Kenya	Perceptions		Community perceptions of the status and threats facing mangroves of Mida Creek, Kenya: Implications for community-based management	Mangroves; Kenyan coast; Ecosystem services; Management;Community-based Natural Resource		Owuor et al. (2019)

Asia	Indonesia	Livelihoods	Proceedings Paper	Livelihood of local communities and successful mangrove conservation in East Lombok, Indonesia	Diversity of mangrove fauna; Successful rehabilitation of mangrove species	Agriculture	Idrus et al. (2019)
	Philippines	Climate change	Article	Community-based mangrove management, experience and challenges in the context of changing climate	Climate change; Coastal areas; Community-based mangrove rehabilitation	Biodiversity & Conservation	Pulhin et al. (2017)
	Vietnam	Coastal erosion		Mangrove management, assessment, and monitoring	Climate change adaptation; Mangrove rehabilitation; Erosion protection; Co-management; Integrated coastal area management		Schmitt et al. (2013)
	Indonesia	NGOs		Participatory impact assessment framework to evaluate a community-led mangrove and fisheries conservation in West Kalimantan, Indonesia	Mangroves; Community-based conservation; Impact evaluation; Mud crabs; Participatory methods		Miller et al. (2019)
	Cambodia	Participation		Opportunities and challenges for participatory management of mangrove resource (PMMR)	Participatory mangrove management; Cambodia; Conservation		Nop et al. (2017)
	Bangladesh	Sundarbans		Development pathways for co-management in the Sundarban mangrove forest: A multiple stakeholder perspective	Local people; Forest protection; Governance; Skill-based training; Participation	Business & Economics	Begum et al. (2023)
	Bangladesh	Property rights		Attitudes towards current and alternative management of the Sundarbans Mangroves to achieve sustainability	Co-management; Bangladesh; Demand-side policies; Alternative livelihoods; Sundarbans Mangrove Forest; Property rights	Development Studies	Roy and Gow (2015)
	Bangladesh	Sundarbans		Evaluating social equity in protected areas co-management; Evidence from the Sundarbans Mangroves, Bangladesh	Co-management; inequitable; mangrove; protected areas; social equity; the Sundarbans		Mollick et al. (2022)
	Thailand	Collective action		Outcomes of state vs. community-based mangrove management in Southern Thailand	Biodiversity; Collective action; Ecology; Forest management; Institutions; Sustainability	Environmental Sciences & Ecology	Sudtongkong and Webb (2008)

Asia	Indonesia	Livelihoods	Article	Effectiveness of community-based mangrove management for sustainable resource use and livelihood support: A case study of four villages in Central Java, Indonesia	Mangroves; Livelihood; Ecosystem services; Community governance; Sustainable management; Participatory mapping	Environmental Sciences & Ecology	Damastuti and de Groot (2017)
	Thailand	Perceptions		Factors affecting fishers' perceptions, benefits, threats, and participation in mangrove management in Pak Phanang Bay	Coastal fishers; fishing community; mangrove ecosystem services; ordinal logistic regression; point-centered quarter method		Jumnongsong et al (2015)
	Sri Lanka	Social networks		Management, stakeholder collaboration and social network perspective of mangroves	Forest management; Conservation; Mangrove restoration; Environmental policy; Wetland		Nijamdeen et al (2023)
	Bangladesh	Sundarbans		Evaluating co-management in the Sundarbans mangrove forest of Bangladesh: Success and limitations from local users' perspectives	Co-management; Community; Conservation; Evaluation		Siddique et al. (2024)
	Indonesia	Biodiversity		Effectiveness of community-based mangrove management for biodiversity conservation: A case study from Central Java	Mangrove structure; Macrobenthos; Community governance; Rehabilitation; Demak	Forestry	Damastuti et al. (2022)
	Malaysia	Perceptions		Collaborative management at Matang Mangrove Forest Reserve in Malaysia: An assessment of local stakeholders' view point	Mangrove management; Local-population perception; Stakeholder involvement; Forest management; Socio-ecological system; Ecosystem services		Martínez-Espinosa et al. (2020)
	Bangladesh	Protected areas		Assessing good governance and co-management in protected areas: A case study in Sundarban Mangroves	Sundarbans; Protected areas; Good governance; Co-management; Mangrove forests		Mollick et al. (2022)
	India	Sundarbans		Perceptive insight into incentive design and sustainability of participatory mangrove management: A case study in Indian Sundarbans	Mangrove conservation; Community-based mangrove management; Joint-forest management; Community perception		DasGupta and Shaw (2017)

Asia	Bangladesh	Sundarbans	Article	Is co-management a double-edged sword in the protected areas of Sundarbans mangroves?	Sundarbans; Co-management; Common's tragedy; Inequality and ethics; Ecosophy and ecofeminism	History & Philosophy of Science	Rahman (2022)
	Indonesia	Coastal protection		Effectiveness of community-based mangrove management for coastal protection: A case study from Central Java, Indonesia	Coastal erosion; Mangrove structure; Java; Indonesia; Coastal protection; Community governance; Climate change	Oceanography	Damastuti et al. (2023)
	India	Effectiveness		Application of criteria and indicators in community based sustainable mangrove management in the Sundarbans, India	Conservation; Participatory management; Climate governance; Multi-level governance; Mangroves; Common resources.		Datta and Chattopadhyay (2010)
	Thailand	NGO support		Monitoring community-based mangrove management in four coastal communities along Andaman Sea, Thailand	Community-based mangrove management; Coastal communities; Andaman sea; Thailand		Kongkeaw et al. (2019)
	Bangladesh	Sundarbans		Determinants of participation of mangrove-dependent communities in mangrove conservation practices	Alternative livelihoods; Co-management; Demand-side policies; Property rights; Sundarbans mangrove forest in Bangladesh		Roy (2015)
	Indonesia	Ecotourism		Collaborative-based mangrove management model for development of marine ecotourism in Lembar Bay, Lombok, Indonesia	Mangrove ecosystem; Marine ecotourism; Sustainability index; Collaborative management	Science & Technology - Other Topics	Sukuryadi et al. (2020)
Latin America	Suriname	Perceptions		Including local knowledge in conservation planning: the case of the western coastal protected areas in Suriname	Torsten Krause; Mangrove forest; Traditional knowledge; Conservation planning; Multiple use management area; Local community engagement	Biodiversity & Conservation	Djosetro and Behagel (2024)

Latin America	Cuba	Climate change	Article	Perception of mangrove social-ecological system governance in southeastern Cuba	Ecosystem service; Ecosystem services; Mangroves; Management; Mitigation; Climate change; Adaptation; Nature-based solution; Citizen participation	Environmental Sciences & Ecology	Portorreal et al. (2024)
	Ecuador	Common property resource		Cockles in custody: Role of common property arrangements in the ecological sustainability of mangrove fisheries in the Ecuadorian coast	Mangroves; *Anadara tuberculosa*; *A. similis*; Artisanal fisheries; Collective action; Co-management; Common property; Community-based natural resource management; Ecuador; Sustainability; Social-ecological systems		Beitl (2011)
	Ecuador	Land tenure		Mangrove concessions: An innovative strategy for community mangrove conservation in Ecuador	Mangroves; Forest concessions; Deforestation; Participatory management of mangroves; Shrimp industry; Black arch production		Rodríguez (2018)
	Ecuador	Mangrove governance		Participatory governance for mangroves in Ecuador	Mangrove conservation; Participatory management; Climate governance; Multi-level governance; Common resources	Forestry	Iñiguez-Gallardo and Fausto López-Rodríguez (2024)
North America	St. Lucia	Livelihoods		Conservation and sustainable livelihoods: Collaborative mangrove management in St. Lucia	St. Lucia; Management; Participation; Wetlands; Caribbean		Geoghegan and Smith (2002)
South America	Ecuador	Payment for ecosystem services		Does exclusion matter in conservation agreements? A case of mangrove users in the Ecuadorian coast using participatory choice experiments	Payment for environmental services; Local users; Conservation contracts; Conservation incentives; Discretionary access; Preference heterogeneity; Club goods	Development Studies	Maldonado et al. (2019)
Worldwide	Worldwide	Participation	Review	Community-based mangrove management; A review on status and sustainabilty	Coastal indigenous community; Community based mangrove management; Community institution; Alternative livelihood; Sustainability	Environmental Sciences & Ecology	Datta et al. (2012)

Worldwide	Worldwide	Monitoring	Article	Mangrove management, assessment and monitoring: An overview	Mangroves; Planting; Rehabilitation; Restoration; Monitoring; Co-management; Site assessment; Hydrology; Coastal dynamic and protection; Climate change; Ecosystem services; Economic values	Forestry	Schmitt and Duke (2015)
	Worldwide	Wetlands	Review	Governance of coastal wetlands; Beyond the community conservation paradigm	Mangroves; Salt marshes; Participation; Co-management; Livelihoods; Tenure; Power	Oceanography	de Oliveira et al. (2024)

4. Conclusion

Mangrove CM offers significant opportunities for integrating biodiversity conservation with sustainable livelihoods. However, its success depends on addressing persistent challenges, including socio-economic disparities, gender inequities and financial constraints. The growing emphasis on collaboration with governments and NGOs reflects a broader recognition of the need for multi-stakeholder approaches to tackle these issues. Studies such as Damastuti et al. (2022, 2023) and Miller et al. (2019) highlighted the importance of external support and community-led solutions, respectively, in achieving conservation outcomes. These insights underscore the importance of adaptive, participatory and inclusive approaches to mangrove management, offering valuable lessons for future conservation initiatives worldwide.

Mangrove co-management offers opportunities for integrating biodiversity conservation with sustainable livelihoods. However, its success depends on addressing persistent challenges, including socio-economic disparities, gender inequities and financial constraints. The growing emphasis on collaboration with governments and NGOs reflects a broader recognition of the need for multi-stakeholder approaches to tackle these issues, pointing to an emphasis on the co-management aspect of what we can call the 'CBM-CM duality', where initiatives can incorporate, to varying extents, an emphasis on community-led action and a push for the inclusion of further groups in management, including private and state actors. This duality, however, is not without its challenges, and tensions can become evident between community leaders and state-related organizations (Borges et al., *in preparation*).

In synthesizing the findings from the studies, several critical themes emerge that are essential for understanding and improving mangrove conservation through CBM and CM. First, community participation stands out as a cornerstone of successful mangrove management, with local involvement driving both ecological and socio-economic outcomes. However, achieving equitable participation remains a challenge, as marginalized groups, such as women and subsistence users, are often excluded from decision-making processes. Second, governance and equity are recurring concerns, with many studies highlighting the need for inclusive governance structures that ensure fair benefit-sharing and address power imbalances between communities and state actors. Third, external support from governments, NGOs and international organizations plays a pivotal role in providing the necessary funding, technical expertise and institutional frameworks to sustain conservation efforts. Finally, the balance between conservation and livelihoods is a persistent theme, with successful initiatives often integrating ecological goals with socio-economic benefits for local communities, such as alternative livelihoods and income-generating activities. These insights underscore the importance of adaptive, participatory and inclusive approaches to mangrove management, offering valuable lessons for future conservation initiatives worldwide.

Future research should focus on the institutional and socio-political dimensions of CM and CBM, including the impacts of globalization on local communities. Strengthening partnerships among governments, NGOs and local stakeholders is critical to ensuring the long-term success of CM initiatives.

Ultimately, the complexity and diversity of mangrove systems demand adaptive and inclusive approaches to management. By building on lessons from successful initiatives and addressing the limitations of existing frameworks, CM and CBM, either combined in one single management framework or as separate strategies, can serve as a model for sustainable conservation in mangrove ecosystems and beyond.

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Chapter 9

Mangrove responses to challenges of the Anthropocene

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1. Historical roots of mangrove degradation

The circumtropical distribution of mangrove forests and their location along coastlines and within estuaries, placed mangroves at the front rank of European colonial fleets that arrived in these latitudes over 500 years ago. These early settlers and the subsequent waves of immigrants from Europe, viewed mangroves as either a nuisance to colonization, or as a source of rapid and easy profit. Both views brought deforestation and degradation of native mangroves, to a point of near eradication in some sites of geopolitical importance to the colonial powers. In Guanabara Bay, Rio de Janeiro, Brazil, for example, over 50% of the bay's mangroves were cleared in the first 200 years of the arrival of the Portuguese fleet in 1500 AD. Paradoxically, it was the Portuguese in 1760, by way of the first, if not, the very first law to protect and adequately manage mangrove, who recognized the importance of these ecosystem services. The law issued by the then King of Portugal and Brazil, D. José, established penalties for cutting trees that had not been previously debarked, not to fulfill any 'environmental' motivation, but rather to meet the demands from local leather guilds that depended on mangrove bark and its rich tannin contents. In fact, as early as the 16th century, the Portuguese performed methodical studies on the properties of the tannin obtained from the bark of different species of *Rhizophora*, *Avicennia*, *Sonneratia* and *Xylocarpus* from South India, Moluccas and Sri Lanka.

Portuguese interest in the New World mangroves and their products increased following their decreasing supremacy in Asia (Lacerda, 1999). Later, when the Portuguese court moved to Brazil (fleeing the Napoleon army), Prince D. João VI of Portugal and Brazil founded, the first Practical Chemistry Laboratory of Brazil in Rio de Janeiro in 1812. The objective of the Laboratory was to discover applications for the different natural materials in the Portuguese colonies to industry and commerce, particularly to balance the bilateral commerce with China and decrease dependence on imported goods (Rheinboldt and Azevedo, 1955). The very first study by the Laboratory was an alternative process to produce hard soap, to decrease the Kingdom's expenses from importing soda. The process involved the use of mangrove ash as a source of alkali (Lacerda and Santos, 2004).

This view of mangroves as suppliers of services and commodities or a total nuisance to development, has had a few exceptions in the past 500 years. Noticeable were the 'Mangrove People' (Vannucci, 1989), traditional human groups throughout the tropics living in balance with these ecosystems, while directly benefiting from their services and goods. Unfortunately, these traditional populations have suffered pressure from growing production and resource exploitation in most of today's societies and continue to be systematically displaced along with mangrove destruction and degradation. By the beginning of the present century, the situation became so grave that a group of mangrove scientists published a doom alert about a world without mangroves, where extinction of entire forests or over-fragmentation of large stretches of mangroves was a real nightmare (Duke et al., 2016).

Conversion to aquaculture, oil palm plantations and rice cultivation together explained 43% of mangrove loss until the first decade of the 21st century (Leal and Spalding, 2024). The recognition of some globally important mangrove services started changing the tide. A pivotal moment was the 2004 Indian Ocean tsunami that caused the death of over one quarter million people (Dahdouh-Guebas et al., 2005). This extreme event clearly demonstrated the capacity of mangroves to reduce the impact of the tsunami's wave force, minimizing human death and structural loss (Hashim and Catherine, 2013). The recognition of the mangrove's capacity to efficiently sequester carbon triggered different worldwide blue carbon initiatives, seeking natural, cheap and efficient mitigation tools to global warming. This increased public awareness of the ecological significance of mangroves as *life support ecosystems* having important functions, and motivated support for many restoration and reforestation projects with the solid endorsement of NGO's, multi-lateral organizations, scientists and governments (Ferreira et al., 2022).

As a result of this global effort in favor of mangroves, the 2024 Edition of *State of the World's Mangroves* (Leal and Spalding, 2024) highlighted the tremendous advances that have been made on multiple fronts to safeguard these ecosystems. While data compilation corroborates previous findings of declining rates of mangrove loss since 2010, the IUCN (International Union for Conservation of Nature) Red List of Mangrove Ecosystems (Ximenes et al., 2023) reported that half of the world's mangrove provinces are considered threatened, notwithstanding the decreasing pressure from human activities. These assessments are strongly influenced by both past and projected future losses, but also, more and more, they start considering the threat that climate change poses to mangroves, particularly with respect to sea level rise, changes in rainfall and continental runoff patterns and quantities, and increased frequency and intensity of extreme meteorological and oceanographic events.

The natural rearrangement of the erosion-sedimentation equilibrium and sea level rise (both influenced by climate change), significantly impacts mangrove areas. Impacts are strongest in extreme environments, including semi-arid coastlines, where naturally reduced continental runoff and terrigenous sediment contribution result in relatively small sediment accretion rates (SAR) and accelerate coastal erosion, as demonstrated in different semi-arid world regions (Saderne et al., 2018; Ward et al., 2023; Lacerda et al., 2024). Another significant example of strengthening impact from sea level rise is verified in mangroves from Small Island Developing States (SIDS). A global report by the United Nations (UN-OHRLLS, 2015) suggests that 6 out of the 10 countries most at risk from extreme climate events are SIDS. In the Maldives, for example, extreme increase in sea level, driven by the Indian Ocean Dipole (IOD), provoked significant mangrove dieback impacting on approximately 25% of mangrove-containing islands, in the past decade (Carruthers et al., 2024). The study reported that local sea level rise of 30.5 $mm.yr^{-1}$ easily outpaced local mangrove SAR (6.4 $mm.yr^{-1}$) by five-fold. Increasing strength of the IOD was also associated with mangrove dieback in other islands within the Western Indian Ocean, where the Global Mangrove Watch (2023) reported significant decrease in mangrove area of approximately 1%, such as in Mayotte (6.5 ha) and in Madagascar (422 ha) during the past five years.

In addition, regional assessments highlighted the large variability of impacts. Most patterns of change together with positive feedback from local human activities, have increased the potential for mangrove degradation worldwide, even where these ecosystems are legally protected.

The recent intensification in global climate change drivers has exacerbated the effects of existing local anthropogenic drivers, particularly in extreme environments such as arid and semi-arid coasts (Lacerda et al., 2024). Recently, the emergence of new drivers of the Anthropocene (such as plastics and 'forever pollutants'), suggests the need for sustained efforts to understand their impacts on ecologically important tropical and subtropical coastal ecosystems. Yet, despite (and because) the responses of mangroves to environmental change are still far from understood, we need to act now to protect extant mangrove stands, preventing their clearing and fragmentation and eventually rehabilitating/restoring them using current and developing techniques and available experiences, so that effective functional recovery can take place (Ferreira et al., 2023).

2. An integrated approach to study mangrove response to environmental and human pressures

Most literature deals with the response of the biological component of mangrove ecosystems. However, even this relatively abundant literature lacks a full understanding of the significance of the biological component relative to the structure and functioning of mangroves (Ferreira et al., 2024). Rarely are functional groups viewed in an integrated manner with other non-biological components of the ecosystems. For example, the biogeochemistry of mangrove soil, and therefore its suitability for mangrove root development, is strongly affected by a diverse array of functional groups, from microbiota to components of the benthic macrofauna, particularly crabs. These functional groups respond differently to climate change pressures, therefore exceeding their ecological limits may result in a sequence of events that cannot be understood by simply studying mangrove tree response to a given driver.

A large gap regarding these ecosystems' geochemical and biogeochemical responses to different climate change drivers, still hampers more efficient actions towards the conservation of pristine areas, as well as restoration and rehabilitation of degraded ones. The spatial-temporal regeneration dynamics of vegetation and the distribution of plant functional types after the impact of hurricanes, using small unmanned aerial vehicles (UAVs) and regional climate and local environmental data, show a fast and high recovery rate of about 87% over two years. Essentially, recovery involved an increase in plants with high specific leaf area. Regeneration rate was influenced by groundwater conductivity and waterlogging and was favored by high nutrient availability. Therefore, mangrove plant cover and composition of functional plant types proved resilient to the initial effects of hurricanes and the subsequent changes in conductivity and freshwater conditions (Hernández et al., 2021).

Environmental changes in already extreme environments such as arid and semi-arid coastlines may cause long-term changes in mangroves, with much slower recuperation from impacts. For example, in semi-arid Australia, changes in conductivity expressed by extreme salinity stress led to large-scale mangrove dieback (Dittman et al., 2022). After a peak in soil conductivity, hypersalinity persisted for about a year. In the area adjacent to the hypersaline site, no change in CO_2 efflux from the sediment and above-ground and below-ground carbon stocks occurred, suggesting that mangrove productivity and physiology were also affected in forests adjacent to the immediate dieback zone. The response to extreme hypersalinity in these temperate semi-arid mangroves is similar to that observed in tropical latitudes under semi-arid climates, suggesting they are already close to their physiological tolerance limit, and are therefore more sensitive to environmental change, thus placing them more at risk from extreme hypersalinity regardless of latitude (Lacerda et al., 2024). These findings raise awareness and have relevance for understanding the generality of disturbance effects on mangroves, particularly on the significance of already existing extreme environmental conditions, like in extreme climate regions.

Globally, mangroves occupy the most densely populated areas in the world, the continent-ocean interface. Urbanization allows increasing human density but results in an overall augment in nutrient supply to coastal areas. However, to fully comprehend the key role of nutrient availability to mangroves, it is fundamental to understand resistance and eventually resilience of mangrove ecosystems and their response to drivers of mangrove degradation. Nutrient storage and accumulation in mangroves have been significantly affected by urbanization. Wigand et al. (2021) reported much higher nitrogen (N) storage and accumulation rates in mangrove soils in the 2020s than those estimated prior to 1970, in pace with the rate of human-consumed food N that is exported in wastewater. This strongly suggests a high capacity of mangroves to sequester human-derived N, thus playing a significant role in maintaining water quality in coastal tropical regions. On the other hand, Marins et al. (2020) observed that as mangroves are progressively degraded, their phosphorus (P) accumulation capacity decreases to half the accumulation rates observed in pristine mangroves, suggesting that the nutrient 'filtering' role played by mangroves has a limit and their maximum filtering capacity has already been impacted by excess nutrient inputs.

Biotic factors associated with mangrove structure and community composition have been largely understudied. Some studies give a glimpse into the possible effects of climate change on key fauna and microorganism groups and the consequences for forest community structure (see Ferreira et al, 2024 for a review), but these effects are still largely unknown. Hendy et al. (2022) gave an insight into the influence of rising sea level on forest inundation and its effect on the amount of wood degraded by boring organisms, i.e., biodegraders, which have an important role in nutrient and carbon dynamics through the wood decomposition process. They reported that terrestrial biodegradation processes dominate the high intertidal region of mangroves, whereas in the low intertidal region, marine wood-boring animals belonging mostly to the family, Teredinidae, are the dominant biodegraders. Degradation of large wood debris, mediated by biodegraders, is influenced by inundation time, therefore, rising sea level can affect the amount of carbon stored in and released by mangrove ecosystems. The results of the study also have significant implications for biodegrading guilds because of their importance in accessing mangrove carbon stores.

Approaches that consider the continent-ocean continuum and the processes involved in continent-ocean gradients, are particularly inadequate. In addition, long-term studies covering large geographical scales are scarce and are usually threatened by economic instability of many nations along the world's tropical coasts.

This calls for joint efforts to promote multi-disciplinary research that encompasses the mosaic of natural ecosystems and links watershed processes and ocean sink. These efforts should be aimed towards a comprehensive view of major alterations in the fluxes, transformations and the cycles of substances at the continent-ocean interface, which promote environmental changes that can lead to pollution, eutrophication, oxygen minimum zones and related biological crises.

Priority should be given to studies that assess anthropogenic influences on the interaction between drainage basins and the continent-ocean interface (as well as impacts on continental shelves), influences that may change the cycle and fate of carbon, nutrients and persistent pollutants. The studies should aim at understanding the implications of global changes to ecosystem functioning, conservation and sustainable development, and in particular, the vulnerability of mangrove ecosystems and possible threats to society through food (in)security and human occupations of the coastal zone (Lacerda et al., 2022a).

3. Can sustainable development preserve mangroves?

Globally, over 90% of mangroves are located in developing tropical and subtropical countries, where rates of loss and degradation are high. Mangrove deforestation started in early colonial times, when mangrove forests were estimated to cover 200,000 km^2. As with other tropical forests, about 70% of these mangroves were deforested and converted to other uses (Duke et al., 2017). In addition, prior to the recent reduction in rates of loss, there was an acceleration in the past 50 years that eradicated about 20% (36,000 km^2) of the remaining mangrove forests (FAO, 2007). A more detailed analysis of recent mangrove area loss (1996 to 2016) showed that deforestation and conversion to other uses resulted in a further loss of 9,736 km^2 (6.6%) and degraded an additional 1,389 km^2 (0.95%) of mangroves worldwide.

Mangrove loss rates, however, were highly variable through space and time. Between 1996 and 2016 the global annual mangrove loss was about 0.21% of its total area, whereas in North and Central America and the Caribbean the average rate was 0.36% and in SE Asia, it was 0.29%. Other regions showed lower rates, such as the Middle East (0.23%), Australia/New Zealand and South America (0.14%), West and Central Africa and the Pacific Islands (0.06%) and even a small gain in mangrove area of 0.03% annually in East Asia (Worthington and Spalding, 2018). In general, however, these figures represent a relative reduction in deforestation rates compared to late 20th century figures. Also, notwithstanding the present threats to mangroves, an intensive restoration effort is being witnessed in many regions of the world. Globally, a range of coordinated empirical studies linking drivers of loss and mangrove deforestation to global policy development for mangrove conservation has been put forward, providing some optimism for the future of mangrove conservation (Friess et al., 2019). However, beyond climate change, which can devastate large mangrove areas that will need decades to recover/restore (Duke et al., 2022; Machava-António et al., 2024), the developmentalist paradigm of economic growth at all costs, and recurrent economic crises will put more extractive pressure on mangroves and other wetlands and may negatively impact conservation/restoration programs. This can be seen by fast-growing intensive shrimp farms, which transform shrimp into a commodity at the expense of mangroves and their goods, directly threatening traditional populations (Lacerda et al., 2019, 2021; Ferreira et al., 2022). Carbon credit mechanisms and other greenwashing schemes are by nature, unable to stop mangrove degradation, several times monetizing predicted and extremely variable events (future development of a planted forest and carbon sequestered volumes, for example) that take decades to reach negotiated sequestration volumes, and/or may not even happen (e.g. destroyed by an extreme climate event, a plague, or others). This demonstrates that the essence of these schemes is to treat life support ecosystems, such as mangroves, as market products (Ferreira et al, 2022, 2023).

From a global perspective, the continent-ocean interface is an area that is particularly sensitive to climate change drivers. These global drivers, together with other local anthropogenic drivers, strongly affect the biogeochemistry of the coastal zone. Of high environmental significance is the balance of nutrients and pollutants in mangrove-dominated estuaries, which accumulate these substances through time, particularly those that are non-degradable, such as persistent organic pollutants (POPs) and metals. Mangroves may mitigate or trigger pollutant mobilization in response to climate change, as discussed in recent reviews by Lacerda et al. (2022b), Noor et al. (2024) and Rahman et al. (2024). In general, most chalcophile metals accumulate in mangrove soils as poli-metallic framboidal pyrites, whereas organic pollutants are slowly degraded due to anoxic conditions. High sedimentation rates rapidly bury these substances in soils, avoiding their oxidation and instead favoring their accumulation. Exudation of oxygen by roots fixes Fe, which co-precipitates metals as oxyhydroxides in the rhizosphere. These biogeochemical processes reduce pollutant bioavailability and mobility. Climate change-driven pressures are likely to alter this biogeochemical equilibrium. Sea level rise increases erosion, which favors oxidation of deposited compounds, releasing metals and POP-derivatives to the water column (Japenga et al., 1988; Lacerda et al., 2022b). Pollutants may then adsorb onto suspended particles and can either be re-deposited in the estuary or be exported to continental shelf sediments. Saline intrusion may oxidize deeper sediment layers and also release pollutants to porewaters. Part of the mobilized pollutants may remain in solution complexed with dissolved organic matter, making it highly bioavailable. Unfortunately, disruption of the traditional uses of mangrove goods by humans dwelling in mangrove-dominated coastlines may result from these processes, through increasing contamination of the coastal fisheries that these groups depend upon.

Sea level rise and saline intrusion also trigger mangrove migration both inland and poleward toward their latitudinal limits. This may lead to a change in soil carbon content/accumulation in mangroves, their CO_2 sequestering capacity and thus, their role as mitigators of global warming. This has been demonstrated in mangrove stands thriving in the Atlantic northern limit, in Florida, USA (Steinmuller et al., 2022). Based on soil profile $\delta^{13}C$ compositions, the study observed a range of values reflecting C3 and C4 plant inputs suggesting that shifts in plant taxa occurred, however, mangrove organic carbon burdens were less than or equal to those of co-located tidal marsh habitats. Several explanatory variables (climate, environmental setting, plant physiology and productivity, and duration of encroachment) were proposed to influence soil organic carbon density in mangroves over time – information that is critical for forecasting how continued mangrove expansion might affect blue carbon storage as these habitats evolve. This highlights the need to support long-term monitoring studies at the distributional limits of mangrove forests.

The different approaches and results from comprehensive studies reported recently help draw a scenario of the challenges posed to mangroves facing climate change (Ward et al., 2016; Ferreira et al, 2024). This background information is crucial to make informed decisions concerning management and conservation, as well as to support proper ecosystem function and service provision. Furthermore, as has been highlighted in this book, many mangroves are under different degrees of degradation and many more have been lost. Efforts directed towards their rehabilitation, however, require information on the local or national situation, without which restoration programs are likely to fail. Particular attention should be given to regions that are under-represented in the research sphere, particularly those in Africa, Latin America and parts of Asia, where a large fraction of the human population still relies heavily on mangrove resources.

A visible gap that is evident from the chapters of this book, as well as other recent mangrove literature, is the scarcity of assessments linking environmental impacts on mangroves to societal pressures, particularly those providing detailed accounts of impacts on traditional human populations. Market-oriented solutions, in existence for over 500 years, are not sustainable and have proven to be incapable of stopping mangrove degradation, whereas the displacement and demoralization of traditional populations, or the enticement for them to participate in capitalist global markets by producing a commodity, such as shrimp, has accelerated. As a rule, Marta Vannucci's 'Mangrove People' continue to move toward a point of no-return in their relationships with mangroves and science is still far from knowing how near to a critical threshold they are, thus compromising their place in our fast-changing planet.

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