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ORIGINAL PAPER

New contributions to mangrove rehabilitation/restoration protocols and practices

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Abstract The conservation of extant mangroves and the restoration of the lost forests is extremely important for the Planet. However, there is a scarcity of information about several operational details and methodologies in worldwide mangrove rehabilitation/restoration projects or attempts (RRPA), especially from a more functional approach. Methodological data from case studies can help practitioners' initiatives, projects progress, and/or management measures improvement. In this work, we make a non-systematic review of several mangrove restoration protocols, case studies, and published RRPA, addressing several important aspects raised by them and during our restoration practices in the Neotropics. We address the specificity of mangrove rehabilitation/restoration (R/R) in relation to the regional biotic/abiotic features and the use of experimental approaches, the question of key faunal groups that can be used as indicators of the functionality of restored stands, the mangrove R/R foreseeing the effects of already ongoing climate changes, and finally the challenges posed to mangrove conservation and restoration by the carbon trade and the '*economic developmentalism*' framework adopted by most countries. This work also contributes to update the knowledge of some practical issues scarcely addressed by studies, aiming to improve the effectiveness of RRPA, especially when considering that mangrove restoration and climate change mitigation projects/attempts are expected to increase.

Keywords Restoration methodologies · Ecosystem functionality · Bioindicators · Functional groups · Climate change mitigation · Economic developmentalism framework

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Introduction

The decade 2021–2030 was defined by the UN as the decade of *'Ecosystem Restoration'* and of *'Ocean Science for Sustainable Development',* both aiming to restore life support ecosystems (LSEs) and protect the Oceans to face their current state of degradation. Among target ecosystems, mangroves are extremely important wetlands since they support a lively biologically dynamic frontier at the land-sea interface. Mangroves are salt-tolerant trees that inhabit the mid and upper intertidal coastal fringes (mainly estuaries and deltas) of tropical and subtropical regions (Ricklefs and Latham 1993; Duke et al. 1998). Abiotic factors like rainfall, tidal amplitude, temperature, salinity, soil characteristics (nutrients and oxygen content, grain size, humidity), and biotic factors, like soil microorganism composition and herbivory, are among the main ecological determinants of their development, diversity and high biomass (Smith et al.

1989; Cintrón and Schaefffer-Novelli 1983; Clarke and Allaway 1993; Duke et al. 1998; Sherman et al. 1998; Lee 1998; Krauss et al. 2008).

Mangroves provide high levels of goods (fisheries, timber and other products) and services (biodiversity conservation, coastal protection, climate regulation among others) relative to their extension (Alongi 2002, 2014; Lugo 2002; Manson et al. 2005; McLeod and Salm 2006). Total mangrove ecosystem carbon stocks (vegetal biomass and soil) can reach an average of 856 ± 32 MgC/ha, approximately 11.7 PgC globally (Kauffman et al. 2020) which is, for example, more than 2.3 times the yearly carbon emissions of the USA in 2018 (Ferreira et al. 2022a). Hence, the conservation and eventual rehabilitation/restoration (R/R) of these forests would keep locked in the ecosystem a great amount of carbon that would otherwise accelerate warming if released to the atmosphere. Therefore, their preservation should motivate global efforts.

However, despite a slight reduction in forest loss rates mainly in the Americas, Africa, and Australia (Hamilton and Casey 2016; Friess et al. 2019), mangrove clearing and fragmentation continues, mainly in Southeast Asia (Bryan-Brown et al. 2020; Goldberg et al. 2020; Jennerjahn et al. 2022). Recent research highlights the widespread of indirect impacts on mangroves, such as remobilization of heavy metals deposited in the sediment to the water column, and reduction of ecosystem services like primary productivity, carbon storage capacity, and resistance and resilience to environmental changes and other stressors (Lacerda et al. 2021, 2022). Predictions of carbon stock declines above and belowground (carbon in roots and soil) in any future climate change scenario is a matter of concern (Singh et al. 2022). Therefore, the conservation of all world mangrove areas is a vital mitigation measure to climate change. In addition, several native/indigenous populations use them ancestrally or for subsistence and have the right to their conservation, bringing community engagement and participation in R/R as a key conservation initiative (Glaser et al. 2010; Datta et al. 2011; Borges et al. 2017; Das et al. 2022; Gómez-Ruiz et al. 2022).

Ample mangrove reforestation programs need to be implemented, since in the last four decades only around 2000 km^2 of mangroves have been restored, while 8120 km^2 of lost mangrove forests are yet restorable worldwide. These latter, if restored, could sequester about 69 million tons of atmospheric carbon (equivalent to the annual emissions of 25 million US houses), also increasing small-scale fisheries and coastal protection (Worthington and Spalding 2018). Several programs/projects have rehabilitated/recuperated mangrove stands, from small to larger areas (Field 1998; Primavera and Esteban 2008; López-Portillo et al. 2017). Studies on the progress of these restorations are valuable to learn and reproduce. Unfortunately, however, many other rehabilitation/restoration (R/R) projects/attempts (P/A) failed, in general by lack of a comprehensive biogeophysical evaluation of the site and its history of environmental impacts, lack of medium- and long-term monitoring, as well as exclusion of the pre-degradation traditional uses of mangrove resources, which can result in social conflicts with local populations (Lacerda et al. 2019). Additionally, the impacts of climate change, which are already a reality, were not considered by most R/R programs and replanting experiments, then compromising their long-term benefits. This may eventually reduce the effectiveness of conservation legislation and impair mitigation and adaptation to global climate changes (PBMC 2013; Lacerda et al. 2019; Ferreira and Lacerda 2016).

An effective mangrove rehabilitation/restoration project/attempt (RRPA) needs to rehabilitate/restore (and preserve) the mangrove ecosystem functioning (i.e. ecological processes involving interactions between biotic and abiotic elements like productivity, carbon and nutrient dynamics and sediment dynamics) (Gamfeldt et al. 2008; Aerts and Honnay 2011; Cadier et al. 2020), which leads to more resilience. We use both terms rehabilitation and restoration in the sense of recovery of mangrove ecosystem functionality and biodiversity with native species; 'rehabilitation' characterizes an intervention to set an urgent recovery of mangrove ecosystem functioning with some key species (frequently a previous stage of restoration), while 'restoration' refers to the settling of a more complete functional and biodiverse (and in general more time consuming) mangrove community. The term 'functionality', despite a similar meaning to 'functioning', will be used here when stressing the role of organisms (e.g. ecosystem engineers, functional groups) within the community and the general processes that they influence in the ecosystem (Jax 2005; Bosire et al. 2008; Ferreira et al. 2019b). Functional groups are species assemblages that perform/

provide similar ecosystem processes or functions through resource exploitation (e.g. biogeochemical cycles, seed dispersal, facilitation, bioturbation, etc.) (Blondel 2003). The execution of a RRPA requires an understanding of the physical-biological nature of the mangrove forest and surrounding watershed (Lewis and Brown 2014).

Since the early attempts to rehabilitate/restore mangroves, several protocols and methodologies have described conditions, methods and criteria to achieve mangrove restoration (e.g. Teas 1977; Field 1996; Lewis and Marshall 1997; Stevenson et al. 1999; Lewis 2005, 2009; GNF 2007; Chan and Baba 2009; Primavera et al. 2011, 2012, 2014; Lewis and Brown 2014; Lewis et al. 2019; Herrera-Silveira et al. 2020; UNEP 2020; Teutli-Hernández et al. 2021). The 'Ecological Mangrove Restoration' (Lewis and Brown 2014) and recently the 'Manual para la Restauración Ecológica de Manglares del Sistema Arrecifal Mesoamericano y el Gran Caribe' (Teutli-Hernández et al. 2021) for Neotropics, are extensive manuals assessing diverse issues related to planning, executing and monitoring of mangrove RRPA, covering diverse methodological and practical issues. Additionally, countless experiments/experiences of mangrove restoration have been performed in the last years (see Reviews of Field 1996, 1998; Bosire et al. 2008; Primavera et al. 2012; López-Portillo et al. 2017; Das et al. 2022; Herrera-Silveira et al. 2020; Saunders et al. 2020; Rodríguez-Rodríguez et al. 2021; Su et al. 2021, and references herein). However, more detailed protocols are still needed to restore mangroves aiming a better use of regional biodiversity, use of biological indicators of restoration progress, long-term functioning and, importantly, foreseeing the effects of climate change (Lee et al. 2019; Ferreira and Lacerda 2016). With the increase in RRPA worldwide, and since tree species diversity and community features vary among mangroves of Atlantic-Caribbean-East Pacific (ACEP) and Indo-West Pacific (IWP), several important issues have been raised along different regions.

First, mangrove RRPA need to observe specific local characteristics like physical–chemical setting and connectivity, extant tree species (especially pioneer ones) and animal/microbial biodiversity. Second, they need an experimental design/protocol that can be conducted through an 'adaptive management'. Third, some key faunal groups are fundamental in mangrove functionality and as indicators of restoration state in comparison with reference adjacent forests, like Brachyuran crabs, fishes and microorganisms (bacteria, fungi and algae). Fourth, mangrove RRPA need to consider climatic changes that are already occurring, to gain resilience; also, a revision of the policy of financing mangrove restoration by carbon credits is addressed. There is a lack of information about some operational and methodological details in mangrove RRPA and case studies around the world, and further data could contribute to practitioners' initiatives, projects progress and/or improvement in management measures (Edwards et al. 2021; Friess et al. 2022). In this work we discuss several issues raised along our mangrove R/R practices in the Neotropics (the biogeographical origin of most of the examples and images here) but less addressed by RRPA protocols and case studies' published practices. Hence, this review contributes to improve mangrove R/R execution and management, concerning the expected increase in mangrove recuperation and climate change mitigation efforts.

Issues on mangrove rehabilitation/restoration

Specificity of mangrove rehabilitation/restoration

On a global scale two major mangrove distribution regions are the Atlantic-Caribbean-East Pacific (ACEP) and Indo-West Pacific (IWP), the latter with around four times more tree genera and six times more species than the former (Ricklefs and Latham 1993). Despite restoration methodologies can be partially similar among these two realms, the ecological and architectural traits of forests require the use of specific tree species, experimental settings and practices at a regional level, like continental coasts, great deltas, islands or at a country scale. Locally (a delta or estuary for example), RRPA need to observe specific local characteristics, like tree species, environmental setting of the littoral areas, physical–chemical and climatic features, animal/microbial biodiversity, degradation drivers, system connectivity, and native people's knowledge and historical use of mangrove resources, among others (Duke et al. 1998; Ellison 2000; SER 2004; Krauss et al. 2008; Lewis and Brown 2014; Ferreira et al. 2015; Kodikara et al. 2017; Lacerda et al. 2019).

Defined and clear aims of the RRPA need to be established to select the tree species to use. In every restoration project, the return of the highest possible set of native species is a desired, but not exclusive, goal (SER 2004; Gann et al. 2019). Restoration should include the maximum native species present in each specific region, mainly if it has high tree diversity (and consequently high faunal diversityRicklefs and Latham 1993; Lee 1998). However, depending on the regional setting, monospecific restorations can perform better than mixed species, which on the other hand not always give better outcomes (Su et al. 2021). Indeed, the presence of some key species may already trigger the return of ecological processes that can allow further self-development (Lewis 2005; Ferreira et al. 2022b). In coasts with historically low tree richness (e.g. Neotropics, arid and semiarid coasts, regions with permanently variable sedimentary profiles, extremely muddy soils/sediments, among others) selected pioneer, former or key species (in general with more ample niche requirements) can be used initially (Lewis 2005; Flores-Verdugo et al. 2015; Ferreira et al. 2015; Al-Khayat et al. 2019; Erftemeijer et al. 2020). Such species are fitted to these specific environmental settings, and some of them are key for the establishing of significant emergent properties (productivity, diversity, architecture, resilience) of the forest stands.

In the Neotropics, low tree diversity stands (1–3 species) predominate, and restoration with one or few selected species of faster growth and higher primary production, and/or more able to facilitate other mangroves settling and the return of key biological groups, can be an effective tool to hasten the recovery of forest functionality. In fact with varied outcomes, most restoration projects have used one or two species (e.g. *Rhizophora* and/or *Avicennia*), resulting in an initial recovery of a great portion of native tree diversity of the targeted stand (33 to 100%) and consequently presenting a higher probability of a fast return of functional features and diversity to the ecosystem (Field 1996; Ellison 2000; Toledo et al. 2001; Ferreira et al. 2015; Flores-Verdugo et al. 2015; López-Portillo et al. 2022) (Fig. 1). Successful RRPA have been performed in Colombia, México, USA and Brazil (Elster et al. 1999; Rivera-Monroy et al. 2006; Lewis 2005; Ferreira et al. 2015; López-Portillo et al. 2017; Herrera-Silveira et al. 2012, 2020; Rodríguez-Rodríguez et al. 2021).

Fig. 1 Restored mangroves in Northeast Brazil. **A** Restored *Rhizophora mangle* forest in a muddy soil (Potengi River, Rio Grande do Norte State). **B** Avicennia germinans in a semi-arid experimental area (Pacoti River, Ceará State)

Cosmopolitan trees of the genus *Rhizophora* are the most commonly used in RRPA, due to their conspicuousness, easy propagule collection and planting, rapid growth, resistance to salinity fluctuations, high biomass and hence, high carbon stocking, among others (Field 1996; Proffit and Devlin 2005; Primavera and Esteban 2008; Samson and Rollon 2008; Salmo III and Duke 2010; López-Portillo et al. 2017; Ferreira et al. 2019a; Barnuevo et al. 2017; Pascoalini et al. 2022). For example, monospecific stands of *Rhizophora mangle* are common in the Neotropics, which facilitates their restoration after deforestation. It can fasten organic matter and further detritus accumulation in soil, fueling the detritivore-based food web, and allowing the settling of other mangrove species also by trapping propagules between prop roots (Menezes et al. 2005; Leite et al. 2021; Ferreira et al. 2022b) (Fig. 1A). This accelerates the colonization by important key groups for mangrove functionality and restart of their biogeochemical cycles, like Brachyuran crabs and microorganisms. However, it can present problems facing climate change effects, and like other monospecific reforestations, be more extensively affected by pest outbreaks (Sect. 2.4).

In semiarid climates, the initial restoration with species resistant to hypersalinity and drought like *Avicennia* spp. can facilitate the return of mangrove stands (Toledo et al. 2001; Flores-Verdugo et al. 2015; Al-Khayat et al. 2019). *Avicennia* stands can stock relatively high amounts of carbon belowground (in roots), unlike *Rhizophora* which accumulates considerable amounts of aboveground carbon (Adame et al. 2021) and can be used to ameliorate soil and decrease thermohaline stress to the colonization of other species. For instance, *Avicennia* can colonize hypersaline soils in Northeast Brazil (Fig. 1B), and it is able to initially develop in salt flats where tides flood these areas upstream and landward (Ferreira et al. 2022b). *Laguncularia racemosa* is a pioneer opportunist Neotropical species that colonizes impacted/degraded areas and can be managed to primarily establish in these realms, since their young stands can present the same diversity of groups such as fossorial/herbivore crabs and grazer gastropod snails compared to multispecific tree fragments (Ferreira et al. 2019b; de Aviz et al. 2020). Some RRPA, however, have not been succeeded, either by lack of experimental design, use of young plants instead of propagules, and/or higher susceptibility of *L. racemosa* and *Avicennia* spp. to mortality due to physical, chemical, and biological factors such as sedimentation, desiccation, tides, and crab predation (Delgado et al. 2001; Rovai et al. 2012; Ferreira et al. 2013). Further, influence of herbivory needs to be considered, and defoliation and propagule consumption by insect outbreaks can influence restoration outcomes, thus needing to be assessed in specific regions.

In high tree species diverse mangroves forests like in the Indo-West Pacific (IWP) (Ricklefs and Latham 1993; Duke et al. 1998), resource partition narrowed the niche width of tree species through evolutionary history, which is difficult to reproduce in RRPA, mainly in small areas, and thus far more difficult to restore. Yet, the number of used species is related to the ancestral use of some of them by traditional human groups (Ellison 2000). Around 75% of the world RRPA are in the IWP region, and the use of one or few species has led to unsuccessful outcomes (Ellison 2000; Primavera and Esteban 2008; Samson and Rollon 2008; Salmo III and Duke 2010; Barnuevo et al. 2017; Lee et al. 2019; Gatt et al. 2022). Since higher tree diversity mangroves in the IWP harbor higher faunal diversity, e.g. key faunal groups such as Brachyuran crabs (Lee 1998, 2008), the concern is mainly about the underdevelopment of functionality of these mangroves if restored with low diversity, as well as low resilience and decrease of ecosystem services (Salmo III and Duke 2010; Barnuevo et al. 2017; Lee et al. 2019). In Asia, used species are commonly *Sonneratia, Avicennia* or *Rhizophora*, and some RRPA also used *Xylocarpus*, *Excoecaria*, *Heritiera* and *Ceriops* (Field 1998; Ellison 2000; Das et al. 2022). To start a RRPA, *Sonneratia, Lumnitzera* and *Avicennia*, can be used for initial plantings, since they are pioneer species able to colonize the littoral zone where tidal inundation and waves influence are higher (Friess et al. 2012; Lewis and Brown 2014).

Exotic mangrove species have been used for restoration, but this is not appropriate, since they can naturalize and eventually become invasive species and threaten the native biodiversity (Biswas et al. 2007; Fourqurean et al. 2010; Ren et al. 2010; Wang et al. 2021). Plantations and expansion of non-native *Sonneratia apetala* in China for example, despite their ability as a high carbon storage species, maintain a lower faunal diversity than native forests (Ren et al. 2010; Wang et al. 2021). On the other hand, the forestation of endangered species, like the neotropical *Pelliciera rhizoporae,* have been scarcely tried (Rodríguez-Rodríguez et al. 2020) and need to be addressed.

Sound restoration methodologies

Mangrove restoration needs to be done in ecologically based and ethical ways. It is also guided by principles and standards for the practice of ecological restoration (Gann et al. 2019). It is necessary to stress that the most cost-effective and efficient measure for carbon sequestration management is to conserve existing mangroves, but, unfortunately, protective measures are ineffective or remain underfunded (Friess et al. 2019; Edwards et al. 2021; Ferreira et al. 2022a). An appropriate experimental design is important and before its beginning, the aims of the RRPA (i.e. the management measures) need to be established. Ideally, a complete assessment/diagnosis of the site conditions, such as degradation history, hydroperiod (level, frequency and time of flooding), regional species and soil physicochemical features (salinity, nutrients, topography), need to be considered to support methods and decisions (Teutli-Hernández et al. 2021). Together with these preliminary assessments it is fundamental for the project to have good planning and implementation, sufficient knowledge, effort and resources, comprehension of social context, stakeholder involvement and monitoring (Lewis and Brown 2014; Gann et al. 2019). Yet, mangrove restoration needs to be integrated with serious efforts to protect and rebuild degraded coastal and marine ecosystems, aiming to increase the abundance of keystone species and key habitats and restoring the three-dimensional complexity of benthic habitats of the main Life Support Ecosystem, the Oceans (Duarte et al. 2020).

Early restoration efforts tended to use the historical condition as a management target based on the concept that biota tends to a climax or 'equilibrium'. However, equilibrium state is not common in nature, and the selection of a specific historical condition is not necessarily desirable or even an achievable goal in mangrove restoration (Wyant et al. 1995; Choi 2004; Suding and Gross 2006). This is recognized in the updated definition of restoration from *Society of Ecological Restoration*: '*Ecological restoration aims to move a degraded ecosystem to a trajectory of recovery that allows adaptation to local and global changes, as well as persistence and evolution of its component species*' (Gann et al. 2019). Using sound methodologies, and assuming that multiple recovering trajectories are possible (Choi 2004), restoration can be developed as experiments and used as case-studies to set more specific or general models for future restoration attempts (Ellison et al. 2020).

Sometimes, it is preferable to leave the target area to recover by itself, but this needs a preserved environment (including a regular tidal regime and connectivity), sedimentary areas suitable for mangrove colonization, and a measurably genetic pool from adjacent areas. This can be achieved if there are few financial resources or sufficient time for the ecosystem to recover without human or natural (extreme events) impacts. If mangroves show signals of self-recovery they just need to be protected. In other cases it is necessary to intervene by removing physical–chemical impairments and/or barriers that prevent waterborne propagules to establish and develop (e.g. dykes, deactivated saltworks or shrimp pond barriers, impairments of tidal coverage, deviations of watercourses, cattle breeding, hypersaline or degraded soils, garbage dumps or eutrophication sources) (Ferreira and Lacerda 2016). For example, abandoned salt and shrimp ponds enterprises leave barriers among ponds, impairing natural recovery. Hydrological management to restore tidal reach in such and other target areas is an effective measure in several RRPA (Ferreira et al. 2022b).

When the conditions for mangrove recovery are not appropriate due to degradation or conversion of preexistent mangroves, planting can be a solution, providing that the selection of tree species for restoration is well planned, as well as the goals of a given RRPA, the methodologies and the monitoring plan. If there are areas at risk of human advance (e.g. dumping grounds, herding, human settlements) with potential impairments to the ongoing natural recovery of mangroves, then planting can be used to accelerate the area recovering before the human advance (Ferreira and Lacerda 2016). If a direct intervention through planting is needed, planting success depend on how effective the sediment furnishes nutrients and stability to propagule rooting. Since it is inappropriate to bring sediments from other sites to improve soil conditions to propagule planting, in sites with degraded sediments it may be necessary to wait for natural soil recovery, which depends on hydrology, i.e. tidal flooding extension, prior to perform planting tests.

Defined and clear objectives

The establishing of clear goals/objectives is extremely important in restoration attempts (Field 1998; Ellison 2000; Gann et al. 2019; Teutli-Hernández et al. 2021). The aims can be the conservation and landscaping of the stand, or to recuperate/improve specific goods and services provided by mangroves, both for ecosystem conservation or for the benefit of populations whose livelihood depends on mangrove (Field 1998; Borges et al. 2017). Whatever the goal, the observance of several critical principles and steps is important (Lewis and Marshall 1997; Lewis 2005). Yet, mangroves are part of a diverse coastal landscape, and it is also necessary to consider the adjacent ecosystems (e.g. marshes, coral reefs, grass beds and others) for an integrated ecosystem-based management, and without collateral effects of mangrove restoration over them (Borges et al. 2017). Indeed, coasts and mangroves are mosaics of areas with different successional histories and in constant change (Alongi 2009), so restoration outcomes can affect or be affected by adjacent environments (Sharma et al. 2017). For example, the expansion of afforested mangroves has reduced shorebird populations (some of them endangered) foraging along migratory areas in China, Australia, New Zealand, and Taiwan (Choi et al. 2022). In the Caribbean, mangrove-rich systems have a strong influence on fish community biomass in adjacent coral reefs, mostly by providing refuge for juveniles. In addition to protecting the coast and mainland areas from hurricanes, restoration of Caribbean mangroves could favor offshore coral reefs, since corals connected to mangroves showed recovery evidence after disturbances, unlike reefs without ecosystem connectivity (Mumby and Hastings 2008).

In a world under environmental, political and economic crises, more and more people's livelihoods will depend on resources from natural ecosystems. Mangroves offer livelihood to many people through gathering and fishery of countless resources, thus restoring/protecting the natural goods and services can be a significant societal goal (Gómez-Ruiz et al. 2022). Indeed, one hectare of preserved mangroves in Neotropics may house around 5.1 tonnes of mangrove crab *Ucides cordatus* and yields around 20 tonnes of animal biomass per year including fishes, mollusks, and crustaceans (IBAMA/CEPENE 1994; Rocha Junior 2011). Mangroves play a significant role in coastal protection and sequestering of atmospheric carbon into biomass, thus species and densities more able to decrease tidal/wave energy and increase carbon stock respectively can be preferred. *Rhizophora stylosa*, *Kandelia candel* and *Sonneratia caseolaris* are *s*pecies reported to reduce wave impact, and were used in Vietnam for this purpose, for example (Powell et al. 2011). On the other hand, mangrove restoration can be directed to avoid the human advance over degraded mangrove sites for housing, husbandry, aquaculture or agriculture, alternatively generating outcomes based on better environmental practices for traditional communities (Ferreira and Lacerda 2016).

In regions of extensive mangrove forests, the same forest mosaic at larger scales may exist, and the scale of a RRPA may represent a very small endeavor relative to extensive mangrove areas continually expanding or decreasing naturally or drowning by sea level rise. In northern Brazil mangroves (States of Maranhão and Pará), constant natural changes in estuarine morphology and sedimentary patterns determine extensive areas in a different state of sedimentation. Sediments from the great Amazon basin are the main driver of large and medium-scale changes in coastal configurations and mangrove cover. However, localized RRPA can aim to restore forest fragments to interrupt the process of human occupation, or degradation by interruption of tidal flux by human infrastructure (e.g. roads, ports), and bring awareness and resources to native/traditional populations settlements (Fig. 2).

Despite the planning of the RRPA, unexpected events can impair the restoration process. In a RRPA in a saltwork in semiarid Northeast Brazil (Ceará State), planted *R. mangle* propagules were infected by a defoliating moth that killed around 40% of the saplings in the first year (Ferreira et al. 2022b). Additionally, the sediment retained by the proper *Rhizophora* seedlings shallowed the channels dug to restore site hydrology, thus killing another 40% of the planted seedlings by the return of soil hypersalinity and thermohaline stress conditions after 4 years. So, mangrove RRPA can be developed as experiments and used as case studies to set more specific and better models for future restoration efforts on large areas through an *adaptive management* process, i.e. an '*iterative process of learning-by-doing and decision-making*', to counterbalance the continuous change or uncertainty of environmental (and socio-political) events (Ellison et al. 2020). *Adaptive management* requires regular monitoring of key indicators to determine if the goals of a RRPA are being met, and also clear triggers or decision-points for appropriate intervention and action if the objectives or goals are not being met (Gann et al. 2019).

Hydrological restoration

Often, hydrological features of a degraded area need to be restored. Constructing channels to bring tidal water to the target area is a common management measure when forest tidal connection is impaired by human facilities (e.g. roads, aquaculture ponds), shallowing of natural creeks, or others (Garcés-Ordóñez et al. 2021; Teutli-Hernández et al. 2021; Ferreira et al. 2022b; López-Portillo et al. 2022). In abandoned aquaculture or saltworks ponds, barriers among ponds impair natural recovery by years or even decades, so

Fig. 2 Participation of traditional/native populations in conservation, management, and operationalization of restoration methodologies make them protagonists of the endeavor and improve awareness about the necessity and benefits of mangrove restoration. **A** Deadwood removal from an abandoned area cleared to install shrimp ponds. **B**, **C** Constructing channels for hydrological restoration; the areas had their connection with the estuary interrupted by the construction of roads. **D** Planting *Rhizophora* sp. Propagules by direct insertion of hipocotyle in the soil using parallel extended ropes as guideto plantings (Images from Northeast Brazil, **A**, Potengi River in Rio Grande do Norte State, **B**–**D**, Maranhão State)

the partial or total breaking of such dikes is a primary hydrological restoration measure to allow tidal waters and propagules to get access to the degraded area. While companies are not obligated by law to remove them off after deactivation of activities, the breaking of such barriers in abandoned shrimp ponds by fishermen in the Potengi River estuary (Northeast Brazil) naturally returned mangroves and increased artisanal fisheries captures (Fig. 3) (*Personal communication*). Any hydrological works need to be planned for minimal maintenance, so the further action of biotic and abiotic forces needs to be foreseen to not undo the work (Lewis and Brown 2014).

Natural tidal channels need to be unclogged in case of obstruction. If new channels are needed, the type of sediment must be observed before the channel design and excavation. In general, muddy soil of mangrove areas preclude the use of machinery, being necessary to hire workers. Clay soils are difficult to work; when dry, bulk density is extremely high, making digging and seed planting difficult; when wet, the soils impair workers' dislocation and digging tools operation, despite maintaining more flooding time and humidity for propagules. Sandy soils are easier to excavate, but channels dry more quickly around seeds during low tides because water percolates more quickly. In any case channels need to be planned to consider that sediment carried by estuarine waters will make them shallower with time. It is important to plan where the removed sediment will be placed, since it cannot impair the flooding of the area. Yet, if sediment is polluted, more caution and attention need to be taken to their disposal. A fully analysis of the contaminants present is key to determine sediment fate and final disposal to avoid conditions that might increase contaminant mobility and/or bioavailability, always taking into consideration the sub-oxic or anoxic nature of these sediments. Hired workers need their dislocation planned to avoid excessive trampling of the areas, mainly to avoid the disturbance of crab populations

Fig. 3 Mangrove recovery over shrimp ponds at the Potengi river (Rio Grande do Norte State, Northeast Brazil). At the center-bottom of the image. Two ponds naturally recocvered (≈ 12 years), adjacent to 2 in operation, ta the top-center, 3 smaller ponds with mangrove planted by fisherman and naturally recovered, after partial dyke braeking (≈ 12 years of planted); at the center-right of the image (left river margin), an area of ponds completely recovered by mangrove 20 year after almost total dyke retiring, also by fishermen (Source: Google Image)

(mostly several species of fiddler crabs) which are important in the interaction between planted mangroves and soil (Ferreira et al. 2007). Ideally, sediment removal needs to be done by few tracks, which posteriorly, after deepening, can be used as additional irrigation channels, or removed by carrying away in a boat or similar through the flooded channels (Teutli-Hernández et al. 2021). Eventually, when needed, sediment can be used to create soil patches above the mean water level to propagules establish (Herrera-Silveira et al. 2012, 2020) or to raise planting sites with poor drainage where excessive low level promotes water stagnation and evaporation, which leads to salt accumulation in soil and anoxic conditions (López-Portillo et al. 2022; Dittman et al. 2022).

Ideally, a branching or fishbone design of channels is better (Herrera-Silveira et al. 2020). In function of this, the level/topography of the target area needs to be addressed to take advantage of soil inclination and avoid water stagnation, allowing the input and output of tidal waters in the same way as the tidal frequency. The width and depth of the channels need to be fitted to the size of the restored area, and can have a width up to 3 m while their depth is recommended to be around 1.0–1.1 m (Lewis and Brown 2014; Teutli-Hernández et al. 2021). Channel depth and shape also depend on which and where propagules will be planted, since seedlings need to be out of water in low tides. 'U-shaped' channels (Fig. 4) can be planted at their bottom if sedimentation rate is low, since sediment can bury the seedlings. 'V-shaped' channels (Fig. 5) can be more appropriate at high sedimentation rates since propagules will be planted in the inclined walls, and, if needed, sediment can be more effectively removed of the bottom without damage the seedlings (*Personnal communication*).

Hypersaline tidal flats (or 'apicuns' in Brazil) are common ecotones around mangroves in semi-arid estuaries. These areas have in general less tidal flooding, and are difficult to be recovered by mangroves due to their compacted soil and extreme temperatures and salinities. However, in many places they are in process of natural recovery, e.g. areas where sea level is rapidly rising, thus increasing tidal

Fig. 4 'U-shaped' channels. Planted *Rizophora mangle* seedlings (Pacoti River, Ceará State, NE Brazil)

Fig. 5 'V-shaped' channels. *Rhizophora mangle*, *R. harrisonii* and *R. Racemosa* propagules were planted (Maranhão State, Northeast Brazil). The top right and bottom images show parallel rows of seedlings that are planted taking advantage of the lateral overflow of water in the high tides

flooding, bringing propagules and diluting accumulated salt. Protecting these areas from conversion to any economic activity, pollution or occupation while mangroves recover, can be a cheap and sound methodology for restoration (Ferreira et al. 2022b). Hypersaline tidal flats have been converted into saltworks and shrimp ponds in many tropical regions, and tidal flux restoration can allow the natural recovery bringing free propagules from adjacent mangroves.

Restoration by planting

When the target area is unable to recover by its own natural conditions, if the cause is a lack of propagule availability, it may be necessary to plant, and most RRPA used propagules for this (Lewis 2005). This is a critical phase, since propagules/seedlings need to be planted in areas suitable for mangrove establishing, considering that they will be subjected to the same ecological processes (herbivory, erosion/deposition of littoral areas, burying, unfavourable soil conditions, and desiccation, among others) as natural ones. Inappropriate species and/or site selection are common causes of failures (Lewis 2005; Primavera and Esteban 2008; Wodehouse and Rayment 2019). Thereafter, natural propagule recruitment is an extremely important shortcut to ensure restoration success.

There are three forms of hand planting: direct planting of propagule in the soil, rearing propagules in nursery and planting the seedling, and transplanting of wild seedlings. Each method can work in specific settings; if the site is well selected and the spacing of plantings is adequate, planting propagule directly in the soil is in general the most successful approach (Ferreira et al. 2007) (Sect. 2.2.4). In the Neotropics, for example, to plant inserting propagule of *Rhizophora* in the substrate (a fact that also happens naturally by their shape when it falls from the tree) decrease propagule predation and increase survival, accounting for the predominance of this genera in the Neotropics (Ferreira et al. 2013) (Fig. 6A–C). Similar planting methods of *Kandelia, Ceriops and Bruguiera* propagules in IWP have been successful.

Fig. 6 **A** Two months old *Rhizophora* sp. Seedlings. Planting of multiple propagules can be made; in this case, when a propagule seems to have died we plant a new at its side (we did not remove the "dead" one since in some case it survives and restart to grow). **B** *Rhizophora mangle* seedlings planted following extended rope guides. **C** Same seedlings of **A**, 4 months after planting. **D** Planted *Laguncularia racemose*. Close to their burrow at the centre of the image, a detritivore fiddler crab. **E** Cylinders made from reused plastic bottled to protect planted seedlings of *Laguncularia racemose*, *Avicennia* sp., and *Rhizophora mangle* (Images from Northeast Brazil, **A**, **C** and **D** Marahão State; **B** Potengi River, Rio Grande do Norte State); **E** Pacoti River, Ceará State

In IWP, target species have been also planted regarding the salinity (high salinity: *Ceriops decandra, Bruguiera gymnorhiza, Xylocarpus mekongensis, Avicennia* spp., *Excoecaria agallocha, Xylocarpus granatum, Aegialitis rotundifolia*; low salinity: *Nypa fruticans*, *Sonneratia caseolaris, S. apetala, Bruguiera gymnorhiza, Heritiera fomes)* and ground elevation (Das 2022).

In the case of other Neotropical species, like *Avicennia* spp. and *Laguncularia racemosa*, the seeds are small, and planted propagules are prone to be consumed by herbivore crabs, carrying by tides, or burying by burrowing crabs. So, it may be needed protection to avoid these issues (for example, cylinders from discarded plastic bottles), transplant from nurseries, or to place/plant in sites without herbivore crabs and/or strong currents, where *L. racemosa* can root and grow fast (Fig. 6D and E). Mangrove red crab *Goniopsis cruentata* and several Sesarmids (Grapsoidea) (Fig. 7A and B) and Ocypodoid *Ucides cordatus* (Fig. 7C) can consume mangrove propagules, far preferring these species instead *Rhizophora* (McKee 1995; Ferreira et al. 2013), with implications in mangrove forest productivity and biomass, both higher when *Rhizophora* predominates. In both mangrove great realms (IWP and ACEP) Grapsoid crabs (mostly Sesarmids, but also species with similar ecological roles) can influence mangrove recruitment and tree species dominance, and consequently significant attributes of the

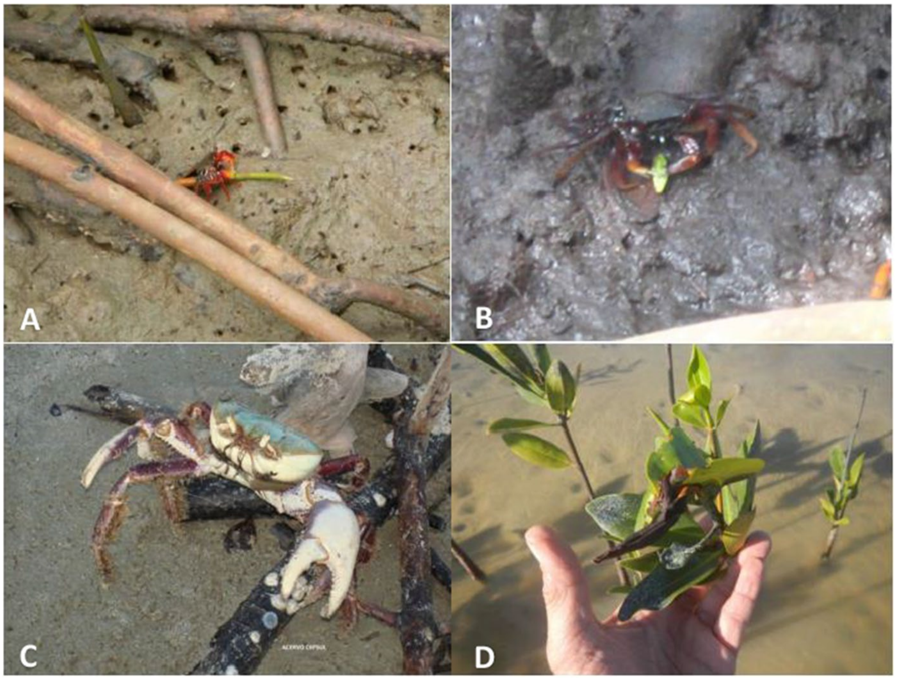

Fig. 7 **A** The red mangrove crab *Goniopsis cruentata* gnawing a *Rhizophora mangle* propagule. **B** The same species eating a *Laguncularia racemose* propagule. **C** The large neotropical mangrove crab, *Ucides cordatus* (Images CEPSUL/ICMBio, Brazil). **D** Apical damage over *Rhizophora mangle* saplings by a parasitic Lepidopera

forest like productivity, architecture, biomass and carbon sequestration levels (Smith 1987a,b; Smith et al. 1989; Ferreira et al. 2015, 2019a). In Malaysian and Australian forests, Sesarmids, beyond *Rhizophora* and *Avicennia*, prefer *Bruguiera* (Smith et al. 1989). IWP's Grapsoids are more diverse than ecologically similar neotropical species (Lee 1998), and despite their influence in natural and restored forests' structure have been not fully assessed yet, they are likely responsible for several unsuccessful plantings attributed to physical–chemical factors. Therefore, crab populations need to be monitored along the restoration process (Sect. 2.3).

Despite already performed with relative success (Toledo et al. 2001; Chan and Baba 2009; UNEP 2020; Das 2022) transplanting nursery reared or wild seedlings ('wilding') in general represents a stress for the plant that most do not support, but this depends on the plant species (Primavera and Esteban 2008; Primavera et al. 2014; Flores-Verdugo et al. 2015). Nurseries have been used for *Bruguiera*, *Rhizophora*, *Ceriops*, *Aegialitis*, *Sonneratia*, *Xylocarpus*, *Nypa* and *Avicennia* in some IWP RRPA (UNEP 2020; Das 2022). Anyway, care should be taken when considering applying resources in large scale nurseries, also a labor-intensive attempt, that can eventually be vain (Lewis and Brown 2014).

Several herbaceous plants, associated to mangrove ecosystems, like *Batis maritima*, *Sesuvium portulacastrum*, *Distichlis spicata*, *Eleocharis* sp., and *Sporobolus* sp. facilitate the growth of *L. racemosa* and *Avicennia* sp. seeds between them, so it is possible to sow these propagules throwing them among these plants, to promote their growth while protecting them from herbivory (Milbrandt and Tinsley 2006; McKee et al. 2007; Teutli-Hernandez et al. 2019). Planting mangrove propagules in clumps (nucleation) have rendered good results and have been shown to increase survival and growth rates of planted mangrove like *Rhizophora* and *Avicennia* seedlings (Huxham et al. 2010; Kumara et al. 2010; Herrera-Silveira et al. 2020). After an assessment on site suitability, planting patches of different species in an estuary/delta is an ecosystem approach that can be used (Su et al. 2022).

Lastly, propagule collection is better (if possible) made in the surroundings of the target area(s), usually in places where currents pile them up. Avoiding picking them from the trees and using the naturally shed ones, not only to ensure some genetic variability but also to warrant that they're mature. Before planting, they need to be carefully inspected to avoid propagules with fungus, crab damages or dried up (Ferreira et al. 2007; Das 2022).

Planting density

Planting density needs to resemble natural spacing in surrounding similar forests, but considering that the density we see is determined by ecological processes like herbivory, competition, change in estuarine sediment settings/profiles, and others. With higher density, self-thinning (density-dependent) process resulting from competition (theoretically higher among conspecific trees) exacerbates, and mortality increases, potentially wasting efforts. Lower densities, like the adjacent forest, are in general indicated for restoring natural forest ecosystem and landscape structural features. However, facilitation among mangroves have been proven, and high-density plantings of mono- and multi-specific tree species have been succeeded in both the Neotropics and IWP realms (Huxham et al. 2010; Kumara et al. 2010; Ferreira et al. 2022b) (Fig. 6).

Since mangrove planting events in general involve local handwork, they need to be optimized so the maximum number of propagules are planted in the restricted time window (usually in rainy seasons, depending on tree species) of propagule release. This trade-off can be partially solved, again, by establishing clear aims for the RRPA. Species density and diversity need to be based on previous scientific and native knowledge with respect to regional mangroves if the aim is to restore the ecosystem for conservation. In addition, since mangroves provide several goods and services for humans, restoration can consider this and find solutions to contemplate both, the ecosystem recovery and its local uses. This deserves caution, however, since an RRPA can plant higher densities seeking to restore/increase forest biomass, crustacean and/or mollusk harvests, coastal protection, wood extraction, fish habitats, and even landscaping purposes, but could promote processes that impair the forest functions or resilience, like excessive sediment accretion or decrease in macrobenthic diversity due to high stem density (Moscatelli et al. 2000; Kamal et al. 2017; Pan et al. 2021).

For instance, in some RRPA at the Neotropics we set highly dense propagule plantings (0.5 m among them (around 4.14 propagules/m^2) of *Rhizophora mangle*, aiming to quickly restore sediment retention, carbon sequestering and stocking, primary production, and/or habitat for animals. We allowed natural processes like self-thinning, herbivory, and facilitation, and obtained biomass values close to older *R. mangle* adjacent stands, and fast return of key faunal groups (Ferreira et al. 2019a). On the other hand, high mid-term roots and stem density, impaired the displacement of crab harvesting fishermen, protecting crabs from overfishing, and giving time to raise fishermen's awareness on the necessary management and resource conservation. Higher propagule densities (0.3 m among 3 plant clusters (around 9 propagules/m) were performed along channels for fast recovery of saltworks on originally converted mangroves, aiming to improve natural propagules retention and fixation, increasing the system organic matter content to fuel the detritus-based trophic chain, and minimizing soil hypersalinity through the increase in populations of soil burrower fiddler crabs (Ferreira et al. 2022b). Lower densities (1 propagule/m^2) can be used to set a more natural tree density, allowing low shade-tolerant species (like neotropical *Avicennia* and *Laguncularia*) to establish.

In the Caribbean, clusters of 10 (multi- or mono-specific) propagules/m^2 or more have been successfully used as 'dispersal centers' planted in raised patches above the water level in permanently flooded areas (Herrera-Silveira et al. 2012, 2020; Teutli-Hernández et al. 2021). Densities of 0.3 propagules/m^2 in multi-specific plantings and high densities up to 8/m^2 for monospecific plantings were performed successfully in IWP (Huxham et al. 2010; Kumara et al. 2010). Spacing of 2 m between seedlings was also used in SE Asia to protect coasts while using 0.1 m among seedlings was in general unsuccessful (Chan and Baba 2009). In multi-specific plantings in India, *Exoecaria* sp. seedlings were planted at 2.5 m apart, followed by lines of *Heritiera fomes*, *Sonneratia apetala*, *Xylocarpus granatum*, *X. mekongensis*, *Rhizophora mucronata* and *Avicennia* sp. 1 m apart. In the lower intertidal portion, *Avicennia officinalis*, *A. alba*, *A. marina* can be planted, and landward, *Ceriops tagal*, *S. apetala*, *X. granatum* and *X. mekongensis* 2.5 m apart. *R. mucronata* and *Bruguiera gymnorhiza* can be planted at the side of creeks (Das 2022).

Finally, planting in rows facilitate counting when monitoring. Parallel ropes tied to stakes can be extended (always avoiding excessive trampling) and marked in every point where a seedling will be placed (Ferreira et al. 2007), or in a checkered design, planted under every crossing.

Assessment of R/R progress through biotic indicators

One of the fewest addressed issues in mangrove restoration is the assessment of RRPA progress through biotic indicators (Bosire et al. 2008; Lewis and Brown 2014). Colonization of restoration sites by both mobile and non-mobile fauna can be a rapid process if there is some connection with healthy mangroves as gene sources, and equivalent populations of mangrove fauna in both natural control and restored mangrove sites can be found within 5–10 years after restoration (Bosire et al. 2008; Lewis 2009; Dahdouh-Guebas and Cannicci 2021) but diversity recovery can take two decades in the case of deactivated aquaculture ponds, for example (Zhang et al. 2021). From a functional point of view, ecosystem's functional diversity is a more relevant component than taxonomic diversity (Naeem 2006). Mangrove invertebrate fauna shows extreme low functional redundancy, so risks of functions collapse by changes in key species composition are higher (Cannicci et al. 2021). Collapse risk likely increase on less diverse Neotropical mangroves through overfishing (e.g. over the ecologically significant crab *Ucides cordatus*), which places these forests as priority in conservation and restoration efforts.

Some key faunal groups are fundamental in mangrove functionality and can be used as indicators of restoration status in comparison with reference adjacent forests. In this sense, the proper tree return is an indicator of mangrove recovering (Thornton and Johnstone 2015), and the presence of growing trees in a restored stand promotes the return of groups that depend on the physical presence (diversity and density) of trees (like insects, crustaceans, fishes, birds, microorganisms, etc.). However, to really consider an organism/group of organisms as true indicator(s) we will focus on their functional role as significant 'gears' (or 'puzzle pieces') of the 'normal' ecological processes of mangrove community development/shaping, and whose presence indicates that such processes are occurring. For example, the macrozoobenthic community contributes to ecosystem functioning and stability and has been positively correlated to mangrove forest tree richness, architecture and biomass (Ellison 2008; Lee 2008; Ferreira et al. 2019a,b). Several ecological processes that promote mangrove growth (and so forest architecture) depends on the microbiota composition of the sediment (El-Tarabily et al. 2021; Farrer et al. 2022). Juvenile and adults of several fish, crab and shrimp species depend on soil burrows for refuge (Barletta et al. 2000; Kristensen 2008; Ferreira et al. 2019b; Lira et al. 2021) with implications in sediment architecture and root development, and thus on microbiota composition.

Brachyuran crabs

Recent studies show Brachyuran crabs as one of the most ecologically significant key groups, since through differential seed consumption and soil bioturbation these 'ecosystem engineers' can determine tree species that establish, and so architectural features and biomass of the forest, which in turn can influence infauna diversity and carbon cycle (Alongi and Christoffersen 1992; Smith et al. 1989; Ferreira et al. 2019a; Barbanera et al. 2022). Hence, the presence of specific crab species/groups in restoring stands indicates that significant biogeochemical processes and specific tree establishment linked to sedimentary processes and bioturbation are occurring. For example, the presence of some Grapsoid crabs indicates tree presence since they need shade to avoid thermal stress, but, mainly, shows that mechanisms of ecological filtering of establishing tree species through selective propagule predation are occurring, with ecological implications (forest diversity and hence architecture, primary productivity and biomass) for developing restored forest. Young plantations are firstly colonized mainly by Ocypodoid crabs whereas Sesarmid (Grapsoidea), omnivore crabs in general dominate mature forests (Macintosh et al. 2002; Ashton et al. 2003; Thornton and Johnstone 2015). The colonization of hydrological restoration channels by specific fiddler crabs (Ocypodoidea) indicates that tidal coverage is reaching restoration areas and bringing their phytobentic food, and hence fueling the detritus-based carbon cycling processes, as well as organic matter deposition and substrate oxygenation (Kristensen 2008). High density of large crabs has been reported to be associated with an increase in forest recover in south Asia (Thornton and Johnstone 2015).

Fishes

Fishes are another functional group that can be used as an indicator (Lewis and Gilmore 2007;

Arceo-Carranza et al. 2016; Hernández-Mendoza et al. 2022). Juvenile fish (and shrimp) species are known to be dependent on structural complexity for refuge; hence, dominance of juvenile fishes of species that use mangroves as sites of refuge, growing and feeding in restoring areas indicates that mangrove is recuperating ecological processes (Primavera and Agbayani 1997; Laegdsgaard and Johnson 2001; Arceo-Carranza et al. 2016). Resident species that are strongly dependent on mangroves, since they reproduce and feed primarily in the forest, typically consists of small fishes with highly variable diets, like cyprinodonts, gobiids (which include burrowing mudskippers in IWP), fundulids, rivulins, poeciliids and eleotrids in the tropical western Atlantic, most of which find shelter in crab burrows and other crevices in the soil among the roots (Barletta et al. 2000; Lewis and Gilmore 2007; Lira et al. 2021). Their presence indicates, like crabs, influence in physical processes in the sediment (e.g. bioturbation), influencing in soil structure and likely in tree root development. *Anableps* sp. (Anablepidae, Cyprinodontiformes) can be useful as indicator of operation of hydrological restoration channels at sites in north Brazil (*Personal communication*), since they enter mangrove creeks and intertidal channels to feed on Grapsoid crabs and intertidal algae like *Catenella* sp. (Brenner and Krumme 2007) showing that restored hydrological flux is allowing return of mobile nekton (Lewis and Gilmore 2007).

Microorganisms

Microorganisms are an extremely important group of organisms in mangrove ecology and functionality (Holguín et al. 2001; Bashan and Holguín 2002; Chen et al. 2016; El-Tarabily et al. 2021; Farrer et al. 2022). In Australian mangroves, bacteria and fungi can reach around 90% of the total microbial biomass, which also includes algae and protozoa (Alongi 1988; Bano et al. 1997). In addition, microphytobenthos (such as microbial mats dominated by Cyanobacteria) contribute to significant amounts of buried carbon in mangrove forests and in hypersaline tidal flats (Brown et al. 2021). Microorganism's complexity increases throughout mangrove plant settlement and development, so they can be used to categorize/identify successional stages during recovery (Chen et al. 2016). For example, since mangrove trees can oxidize the soil by transporting oxygen via roots, edaphic changes induced by developing mangroves can influence the proliferation of certain groups of bacteria in the rhizosphere (Sherman et al. 1998; Holguin et al. 2001), which can indicate restarting of ecological processes promoted by root development. In mangrove sediments, high bacterial activity match with plants presence (Holguin et al. 2001).

Gastropod snails

Gastropod snails, although not strongly associated to community shaping, are indicators of healthy (or developed) mangroves in IWP stands (Basyuni et al. 2022; Nehemia et al. 2017). Gastropod assemblage patterns change as stands grow older in experiments in the Philippines and Neotropics, and some ellobiids and neritids are more abundant in mature stands compared to younger stands (Macintosh et al. 2002; Salmo and Duke 2010; Ferreira et al. 2019b).

To our knowledge there are no specific studies on Insect's functional role in mangroves beyond their role as pollinators (but see Murphy 1990). However, insect defoliations and loss of apical buds can potentially reduce reproductive and vegetative growth, reducing reproductive output and hence influencing species recruitment and consequently forest structure (Krauss et al. 2008; Lu et al. 2019) (Fig. 7C). Increase in temperature and other climate change induced drivers can potentially increase the influence of insects on forests function at the regional level (Pureswaran et al. 2018) (Sect. 2.4.2).

Mangrove R/R responses to climate change

The challenge to restore mangroves foreseeing climate change

Climate change itself represents an increasing direct threat to mangroves, whereas indirectly can potentialize impacts from local anthropogenic activities (Gilman et al. 2008; Moomaw et al. 2018; Lacerda et al. 2022). Therefore, countless studies have been recently published on the subject (e.g. Alongi 2014; Godoy and Lacerda 2015; Ward et al. 2016; Kauffman et al. 2018; Ward and Lacerda 2021) and restoration programs need to effectively consider the scenario of climate change for planning RRPA to obtain functional and therefore, resilient forests. Since mangroves are extremely important planetary Life Support Ecosystems (LSEs), their conservation facing climate change need an 'ecocentric' focus, apart from guaranteeing the maintenance of several goods and services for traditional inhabitants.

Since climate change is already on course (IPCC 2022) mangrove R/R needs to regard the geographical and environmental settings of each site to choose the methodologies with greater chances of success, earliest ecosystem functioning return, and with highest chance to recover or capacity to dampen effects from climate change, i.e. more resilient. For instance, the low tree diversity of Neotropical and West African mangroves, where most stands are predominantly formed by *Rhizophora mangle* (or their congeneric *R. racemosa* and *R. harrisonii*) may allow the restoration through planting of these same species. However, since mature *Rhizophora* lacks epicormic resprouting, in localities where the frequency and/or intensity of great storms are increasing or predicted to increase, a stand of these trees could be seriously damaged and may not recover (Villamayor et al. 2016; Fickert 2020). Though this suggests that in regions prone to extreme weather events like cyclones it is better to initially restore including also non-*Rhizophora* species, the Rhizophoraceae dominate the region with the high cyclone frequency, the Northwestern Pacific (Peereman et al. 2022). Over time, cyclones seems to lead to the development of short-statured forests (Simard et al. 2019) more resilient to cyclone impacts, and with relative higher carbon sequestration in following years, despite reduced aboveground carbon stocks (Friess et al. 2020). Several Asian countries are restoring mangroves in critical areas to protect coasts from storm events (Das 2022). Species like *Rhizophora stylosa*, *Kandelia candel* and *Sonneratia caseolaris* can be used directly to reduce increasing wave impact (Powell et al. 2011). High stem density seems to contribute to resistance to storms (Peereman et al. 2022), so dense reforestations can be a good strategy. Laboratory and field observations estimated that around 100 m of a mangrove belt (≥ 0.11 *Rhizophora* trees/m^2) can mitigate up to 85% of wave height, similar even with different tree arrangements, suggesting the attenuation efficiency of both natural and replanted forests (Mazda et al. 1997; Chan and Baba 2009; Hashin and Catherine 2013).

Climate change introduces more variability in coastal configurations, drivers like sea level rise and reduced annual rainfall, promote erosion and saline intrusion in estuaries. Despite this latter driver can accelerate mangrove expansion and landward migration, for example in the extensive mangroves situated at the Amazon mouth, erosion of mangroves at the river mouths and in subsiding deltaic islands can counterbalance this expansion locally (Godoy and Lacerda 2015; Lacerda et al. 2019). Dynamics of estuaries and coasts driven by these factors (saline intrusion and erosion) require that restoration efforts can combine areas that need to be restored with others undergoing self-recovering, which need to be protected. At species rich Sundarbans Forest, for example, saline intrusion can decrease functional diversity and carbon storage of the forest by affecting less saline resistant species, which are the same that stock high carbon content (Rahman et al. 2021). Consequently, avoiding occupation/use (home, facilities, aquaculture, etc.) of salt flats and high intertidal fringes in estuaries can be a way to ensure grounds for natural or assisted mangrove establishment when sea level rise reaches these areas (Ferreira et al. 2022b).

High-density plantings can promote substrate elevation to counterbalance sea level rise (Kumara et al. 2010; Ward and Lacerda 2021). For instance, the 'dispersal centers' high density planted patches above water level (Herrera-Silveira et al. 2020; Teutli-Hernández et al. 2021), can be a solution in mangroves predicted to be more affected by sea level rise, like mangroves of North and Central America respect to South American ones (IPCC 2022). In coasts with increasing wave energy, the setting of breakwaters can stop erosion and facilitate the sediment and/or propagules deposition, and further mangrove natural or assisted settling, using them as natural barriers. Wooden structures can be used to trap seeds and promote their establishment. The use of natural materials like bamboo, tree branches, woodstakes, etc. (Suripin et al. 2017; Nguyen 2018; Winterwerp et al. 2020) is recommended but avoiding excessive damage over other natural forests.

Conversely, sediment carried by rivers after high rainfall events can bury estuarine mangroves. Hence, the planting in patches and/or the use of species of rapid growing and plasticity to adapt to sediment accretion is essential if restoration is proposed in these regions (Thampanya et al. 2002; Forouzannia and Chamani 2022). Recently it has been observed that events of sea level subsidence driven by ENSO

(El Niño-Southern Oscillation) can affect extensive areas of mangrove (Duke et al. 2022). Countries need to plan massive restoration actions (including propagule/seedlings bank areas) to prevent irreversible degradation in such coasts and adjacent ecosystems. Climate-driven increase in temperature and consequently thermohaline stress in semi-arid or arid areas, lead to prioritize the use of native mangroves able to establish in such conditions, like *Avicennia* species, as addressed in Sect. 2.1). To rehabilitate specific or representative coastal areas connecting them by 'forest corridors' through contiguous Countries would be a form of promoting and organizing multinational efforts to face climate change effects, but this needs, beyond political will, extensive information about the target areas and coordinated efforts among scientists (Lacerda et al. 2019).

Influence of climate change over mangrove architecture/functionality through key faunal groups

An important but still unexplored issue is the indirect changes promoted by climate change over mangrove development and/or architecture through key faunal groups (e.g. ecosystem engineers), leading eventually to functional alterations. This knowledge is important to plan RRPA in the coming times.

Despite Brachyuran crabs are resilient to changes in the forest, the effects of climate change over these ecosystem engineers could affect their richness and distribution. For example, damage in forests by storms or flooding can potentially affect crab assemblages regionally, changing propagule recruitment patterns mediated by Sesarmids. Increased flooding can affect fiddler crabs (Ocypodoidea) distribution with implications in bioturbation patterns, affecting sediment bacterial communities linked to organic matter composition and nutrient and pollutant availability, and hence biogeochemical processes (Kristensen 2008; Booth et al. 2019). Thus, the composition of Brachyuran community needs to be considered when planning mangrove restoration, as we stated in previous sections (Ferreira et al. 2015, 2019a).

Insect herbivory can be important in mangroves and can cause events of heavy defoliation (See Review of Cannicci et al. 2008). In a RRPA in the semiarid of northeast Brazil, severe damage over planted saplings by a Lepidopteran infestation, impaired the restoration effectiveness (Ferreira et al. 2022b). Global and regional increases in temperature can change herbivore and wood boring insect assemblages, and thus intensifying defoliation events or heavy leaf consumption, mainly over young, restored mangroves, but this may be significant on mangroves at their latitudinal limits (Pureswaran et al. 2018). Insect herbivory can change tree diversity and forest architecture, leading to changes in live and dead biomass and, hence, in the aboveground carbon stock. An increase in inundation period, an observed phenomenon linked to climate change (Lacerda et al. 2020) can increase the effect of wood-borer Isopods. These Crustaceans can affect the development and even survival of specific mangrove trees with similar consequences (Perry and Brusca 1989; Svavarsson et al. 2002). Increase in inundation period can also increase Teredinid molluscs biodegradation, which can affect the amount of carbon stored in, and released by, the forest (Hendy et al. 2022). Depending on the scale of these effects they can impact mangrove stands and together with climatic stressors contribute to forest degradation, and therefore, should be considered in RRPA.

Coastal waters acidification and increase in temperature can impair restored mangrove development, through changes in nutrient–plant relationship and several other ecological processes mediated by microbiota in the mangrove sediments (Bashan and Holguin 2002; Lovelock 2008).

Impairment of advances in mangrove RRPA under the 'Sustainable Development' paradigm

Despite optimistic previsions about mangrove conservation/restoration (Friess et al. 2020; Buelow et al. 2022), country's 'developmentalist' framework strikes frontally the required urgent measures to counteract mangrove and other wetlands degradation. While the economic growth/development paradigm keeps leading countries' actions, the environmental degradation by the extraction of more resources than the Earth can replace will continue. Indeed, no economic development is sustainable for the Planet (Ferreira et al. 2022a). While mangroves continue being pressed landward by sea level rise, and seaward by land conversion to many human development facilities (e.g. ports, tourism, sewage treatment stations, urban settlement, aquaculture), which are pressures estimated to increase in the next decades, difficulties to conserve such forests shall intensify. This calls us to identify the real roots of the problem faced by humanity, and to take the risk to restore mangroves that could be posteriorly devastated to allow more'development' and 'jobs creation', or drowned by sea level rise, caused by global warming reinforced by increasing fossil fuels use. At the same time, huge money spent on weapons deviate resources that would be invested in wetlands and Ocean restoration. For example, to extend protection and restoration to 50% of oceans (the most important Life Support Ecosystems—LSEs) by 2050 is possible, allocating US$ 20 billion/year (Duarte et al. 2020), around one-fiftieth of the yearly spendings wasted in military activities by USA and China together (Ferreira et al. 2022a).

Mangroves need to be restored and protected to maintain an extremely important LSE, and this is a significant per se reason. The monetization of nature can be useful to explain the value of environmental services to decision makers (Ferreira and Lacerda 2016); however, it seems to us as a great trap when conditioning forests' restoration to their funding by 'carbon credits' allowing some buyers to comfortably continue polluting or overexploiting natural resources (Ferreira et al. 2022a; Monjane et al. 2022). The 'investors' interested in the blue carbon market trade to finance ecosystem restoration as 'sustainable development' solution [e.g. BHP (mining & petrochemical), Australia, USA, UAE] (Macreadie et al. 2019), are heavy and permanent planetary GHG ('Green-House Gases') emitters together with other Countries, Trusts, and economic activities, like the European Union, China, Brazil, cattle raising, Mining, and Oil exploitation/trading, due to their high life standards, military spendings and/or nature's overexploitation.

Thus, mangrove planting financing has been used as an excuse to increase GHG emissions by considering only the number of seedlings planted without considering the survival rate or feasibility of selected areas for restoration. Furthermore, the carbon market seems to ignore the time mangrove plants need to develop and sequester significant amounts of atmospheric CO_2 (Lunstrum and Chen 2014), while the decrease in GHG emissions is urgently needed. Indeed, carbon trading, to make sense, need to effectively warrant equal or higher net carbon sink than emission, meaning that an instantaneous carbon amount fixed by a restoration/conservation project, for example, funded by a polluter/GHG emitter enterprise/activity, needs at least to equal the emissions by such enterprise/activity operations in the same time interval. This also would require permanent monitoring of mangrove RRPA for a more effective estimation of carbon sequestration amounts. Several examples in South Asia and elsewhere show that deforestation neither stabilized nor decreased after financial input from carbon credits, which neither face the underlying causes of forest loss nor respect and strengthen the rights and knowledge of forest peoples (Kill 2015; Enrici and Hubacek 2018; Ken et al. 2020). Yet, the volatility of carbon market shows their real character of merchandise (Dong et al. 2022).

Concluding remarks

Several aspects that need to be better addressed in present mangrove RRPA are revised here, and despite being treated separately they are related (Fig. 8). The choice of the tree species for a RRPA needs to consider the regional genetic pool and systems' connectivity, to use the appropriate number of native tree species able to furnish early ecological attributes that facilitate the settlement of other trees and functional groups of organisms. The use of biotic indicators needs to be developed to get comparisons among different restored areas, in terms of the facilitative role of interaction among trees and key faunal groups in the restoration of stand functionality and resilience. The necessary knowledge of the history of the degraded forests contributes with an adjusted RRPA program, but the possibility of contingencies needs to be faced with an adaptive management framework able to correct such problems, since climate change-induced drivers are rapidly changing coastal physiognomy/morphology in several places. Finally, mangrove RRPA yield the conservation of an extremely important LSE and needs to be implemented together with a 'zero deforestation' framework around the world, therefore independently of any compensation scheme of carbon trade that allow environmental damage elsewhere.

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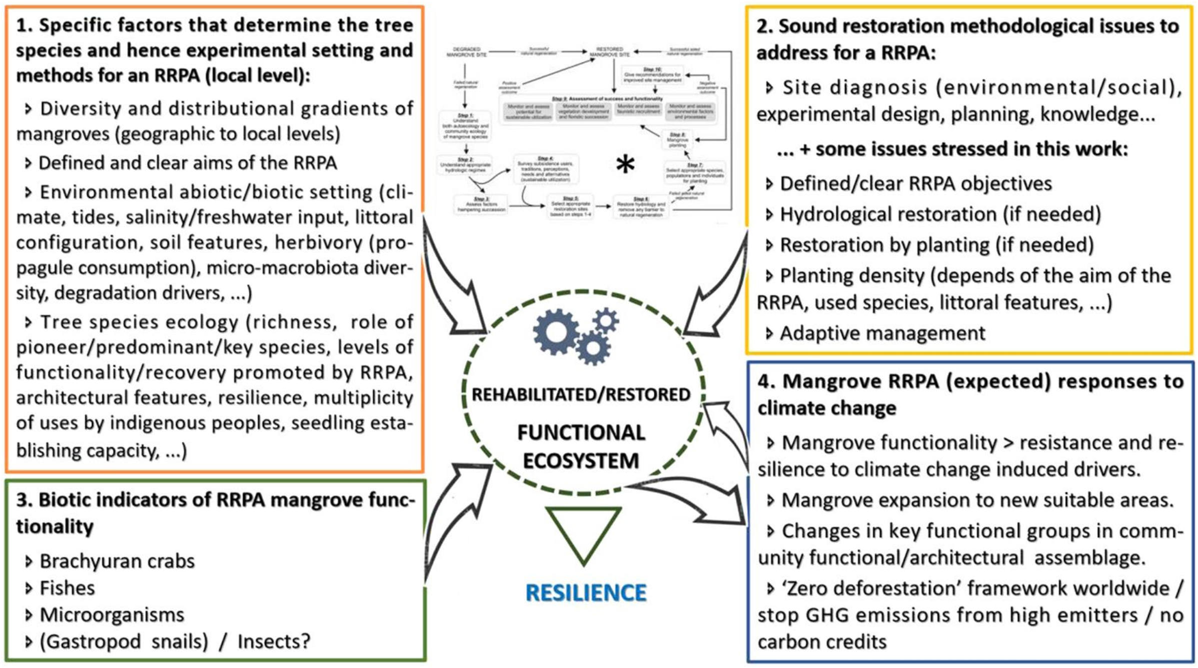

Fig. 8 A summary of the different issues addressed in this work that contribute with the rehabilitated/restored mangroves functionality (i.e. a functional mangrove ecosystem) and further resilience, only considering native species. 1. Specific factors that determine the tree species, and hence experimental setting and methods for an RRPA at local level, several of which are the same that regulates the natural mangrove establishment/recovering; 2. Some aspects of mangrove RRPA methods that were addressed in this work to improve results in terms of mangrove functionality; we only addressed aspects of the steps 5 to 8 of the decision/action mangrove restoration framework of Bosire et al. (2008) (*); 3. Biological indicators of mangrove RRPA functionality/progress; Gastropod snails and Insects potential functional role need more studies. 4. Expected responses of RRPA mangroves to climate change (extreme or not) events; successfully restored mangroves by their time, contribute with mangrove functionality and resilience, so the returning small arrow. For more details about the restoration framework (*) and a full size figure see Bosire et al. (2008)

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Data Availability The datasets generated during and/or analysed during the current study are available from the corresponding author on reasonable request.

Declarations

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

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