DRIVERS OF MANGROVE FOREST CHANGE AND ITS EFFECTS ON BIODIVERSITY AND ECOSYSTEM SERVICES

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Editorial: Drivers of mangrove forest change and its effects on biodiversity and ecosystem services

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Editorial on the Research Topic
Drivers of mangrove forest change and its effects on biodiversity and ecosystem services

Mangroves occupy a global area of 137,600 km^2, roughly the same area as Greece (Bunting et al., 2018). They are one of the most threatened ecosystems on Earth predominantly due to human impacts that have caused over 62% of mangrove loss (Goldberg et al., 2020). Their loss contributed 0.6% of global greenhouse gas emissions related to tropical forest deforestation, despite occupying <0.1% of all land area (Harris et al., 2021). Even with a small footprint, over 200 million people live within 10 kilometers of mangrove forests (Menéndez et al., 2020; Hooijer and Vernimmen, 2021). Mangroves are directly protecting 3.5 million people from the impacts of climate change, including storm surges, flooding, sea-level rise, and erosion (Blankespoor et al., 2017). In addition, mangroves provide habitat to immense coastal and marine biodiversity, offer food and jobs to local communities, and sustain cultural practices and identity. Conservation efforts globally are on the rise, with around 42% of all remaining mangroves being found within protected areas (Spalding and Leal, 2021). However, they may still experience loss related to natural causes and inadequate management (Spalding and Leal, 2021). While this progress spells hope, examples of integrating mangroves into coastal management and policy are still rare. This Research Topic contains a collection of studies (including global assessments, deep dives into issues in eight countries and an author group representing 16 countries), that provide an improved understanding of biophysical, socio-economic, and political drivers of mangrove forest change and their impacts on the provision of ecosystem services. It aims to provide a robust scientific evaluation of the links between enabling conditions—both positive and negative—and conservation and restoration impacts. Thus, identifying successful strategies and sociopolitical drivers that guide cost-effective mangrove conservation and restoration efforts.

When considering management strategies, the saying goes, "you cannot manage what you cannot measure." To that end, many papers in this topic focused on refining mangrove monitoring and carbon pool measurements for informing policy, including carbon and sediment dynamics crucial for assessing climate mitigation potential. For example, Cinco-Castro et al. studied sediment accumulation rates in the Yucatan Peninsula in Mexico. They found high variation in sedimentation rates seasonally and across a salinity gradient. Additionally, a global-scale database compiled from a literature review by Mugi et al. further highlights the importance of mangrove's dead organic matter carbon pool.

A study by Castellanos-Galindo et al. compares multiple assessment approaches where they evaluated mangrove forest structure derived from direct measurement, drone imagery, and satellite-based radar data in a Colombian mangrove. It weighs the efficiency and effectiveness of assessment approaches against the required costs, time, and logistics to produce reliable data. In addition, a national-scale database of Colombian mangrove forest structure provides biophysical information and sets baselines to assess the impacts of changes on ecosystem dynamics (Blanco-Libreros et al.). In Brazil, work by Rovai et al. highlights the importance of mangroves as blue carbon hotspots of global significance. They provide an integrated carbon inventory for Brazil and find the country holds about 8.5% of global mangrove carbon stocks with 15–30% above average carbon sequestration rates—highlighting the importance of protecting mangroves in Brazil. Unfortunately, a second study by Lacerda et al. documents the long-term environmental impacts on mangroves from semi-arid coastal ecosystems, such as salt pan areas and mangrove conversion to aquaculture ponds for shrimp farming. They found that these practices drove direct mangrove loss and lowered productivity, functionality, and services provided by adjacent mangroves and related habitats. Given the increasing demand for aquaculture products, solutions to maintain productivity without expanding production area are crucial.

The effects of land-use changes, such as conversion to aquaculture, on mangrove cover and carbon stocks are well documented (Sasmito et al., 2019; Goldberg et al., 2020). However, the impacts of forestry are less well defined. A study from the Niger Delta in West Africa confirms that wood exploitation in mangrove forests where larger stems are preferentially removed promotes colonization of invasive species like Nypa palm (*Nypa fruticans*) (Nwobi and Williams). These findings emphasize the importance of considering impacts across biophysical characteristics when estimating climate mitigation potential and show that detailed tracking of land-use activities and methods is essential to support national commitments for climate action.

Mangroves' role in climate mitigation has also translated into carbon finance opportunities. Ensuring durable and well-financed outcomes from mangrove conservation and restoration projects is a significant theme for three studies in this Research Topic. Firstly, Pham et al. considers the role of financial incentives in mangrove conservation in Vietnam. They find that contradictory policies, inequitable distribution of power and benefits, and low value of incentives all lead to low levels of compliance. They conclude that while financial incentives can play a role in mangrove conservation, addressing conflicting policies, targets, and governance issues are essential. Secondly, Gatt et al. presents a holistic monitoring framework of key mangrove restoration indicators. Based on >120 restoration reports, they find that studies commonly report on the intervention used and the ecological outcomes, but site conditions before restoration and social and governance outcomes were often missing. Finally, Duncan et al. provides a detailed breakdown of potential return on investment from mangrove restoration and rehabilitation work in the Philippines. They compared natural regeneration vs. assisted natural regeneration—urging caution, as neither approach was highly profitable based on current voluntary market carbon prices.

While addressing governance issues and enabling conditions will be more impactful than focusing only on finance mechanisms—where much of the attention is currently (Pham et al.), competing government agendas challenge mangrove management. For example, Panama's national policy documents recognize mangroves, but economic development is often prioritized over wetland conservation because of a perceived higher return on investment from coastal zone development (Chamberland-Fontaine et al.). The result is conflicting policy objectives, inadequate resources, and institutional structures that struggle to include local communities and stimulate action on the ground (Chamberland-Fontaine et al.). Even when communities are enaged, Chamberland-Fontaine et al. found that often it was the richer and more powerful community members that were engaged, creating a power embalance and a need for a more participatory approach. Further, Scemama et al. demonstrated that different coastal communities in French Guiana provided different perceptions on how they valued mangrove ecosystem services and threats and thus, improved national mangrove management policy should recognize subnational stakeholders.

Actions must be taken now to ensure that mangroves will persist into the future. Government agencies should align and build mangrove management into national conservation, monitoring and climate plans that maximize benefits while allowing local access. Decision-making power should shift to multi-party institutions that allow local communities to lead and promote a collaborative approach to mangrove management.

The private sector should look beyond climate neutrality pledges and expand their investments in these ecosystems as natural coastal defenses for disaster risk reduction, for example to protect supply chains or critical coastal assets. Environmental researchers, policy makers, and practitioners need to do more to build capacity and share knowledge globally. Finally, the public must advocate for their mangroves, reminding elected officials and financial backers of what is at risk and demand their protection.

This Research Topic aimed to capture the variety of drivers and incentives related to mangrove change, however, carbon and climate mitigation was the predominant issue addressed. It is important to continue to expand our understanding around all of the ecosystem service values such that carbon does not overshadow the larger conversation around all the natural capital mangroves provide. For example, further research valuing the high biodiversity of mangroves is needed to promote their protection to support achievement of the goals of the Convention on Biological Diversity's post-2020 Global Biodiversity Framework. The ultimate goal is to promote multiple values held by a diverse range of stakeholders to leverage transformative change and sustainable development (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, 2022).

Equally critical as the activities the world will take to conserve and restore mangroves are leadership and innovation for adopting and amplifying solutions at scale. While the world is waking up to the importance of conserving mangroves, policies and financial mechanisms are only now catching up. However, multinational, cross-sectoral groups such as the Global Mangrove Alliance, Ramsar, and the National Committee on Wetlands champion large-scale, science-driven mangrove protection, sustainable management, and restoration. Collaborative, transdisciplinary efforts will be essential to increase effective and equitable protection and expand restoration. However, this is not enough. Our ability to succeed is relient on active leadership from local communities that take a proactive, instead of reactive, approach. The many values mangroves provide mean that actions to preserve them are not just climate strategies. They are "no-regret" strategies. That message must be communicated and internalized—but more importantly, it must be acted upon.

Author contributions

JH was the lead author with substantive input from DA-B, VH, and SS. All authors provided insight on the aim of the Research Topic, highlighted key points from the papers they reviewed, and contributed to the conclusions and research needs expressed in the editorial. JB as a contributing editor, provided recommendations and comments related to submissions that were integrated into the editorial text. All authors contributed to the article and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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REVIEW
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20-Years Cumulative Impact From Shrimp Farming on Mangroves of Northeast Brazil

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Brazilian mangroves cover about 11,100 km^2 and provide a wide range of ecosystem services. Despite their importance, they are one of the most impacted ecosystems because of combined influences of climate change, pollution, and direct conversion and loss. A major driver of environmental impacts is shrimp farming and this is particularly acute in the semi-arid northeast of Brazil, where mangroves are constrained in a narrow band along ephemeral estuaries that are often impacted by multi-year droughts. Recent changes to Brazilian law, in particular the Forest Code, have weakened protection for mangroves and associated "*apicum*" (salt pan) ecosystems. In NE Brazil, most shrimp ponds are converted from mangrove-adjacent "*apicuns*" rather than the mangroves themselves with periodic hydrological connectivity through dammed channels, allowing the flushing of effluents. As a result, the main impacts on mangroves are typically indirect, because of pollution inputs from shrimp pond effluents and associated loss of ecosystem services including reductions in primary productivity, carbon storage capacity, resilience to other environmental stressors, their efficiency as estuarine filters, and biodiversity and abundance of subsistence use of marine species. Soil damage and infrastructure remaining after shrimp pond deactivation impairs mangrove recovery. This extends the duration of the damage and allows the occupation of degraded areas by other activities that can permanently impair ecosystem function. In this review, we address several aspects of the shrimp culture boom in NE Brazilian, their features and consequences, and the future of mangroves in the region considering climate change and rising poverty. Our conclusions on the practices and outcomes of shrimp farming in mangroves are likely to apply to areas with similar environmental settings, e.g., semiarid regions worldwide, and particularly in the Latin America and Caribbean region, and our findings can be taken into account to improve conservation and management of these forests at the least to a regional scale.

Keywords: aquaculture, eutrophication, deforestation, human impacts, nutrients, blue carbon

INTRODUCTION

Despite a slight reduction in forest loss rates mainly in the Americas, Africa and Australia (Friess et al., 2016, 2019; Hamilton and Casey, 2016) mangrove clearing and fragmentation continues, predominantly within Southeast Asia (80% of direct anthropogenic loss concentrated in Myanmar, Vietnam, Malaysia, Philippines, Thailand, and Indonesia), mainly due to the conversion of mangroves for aquaculture and agriculture [UNEP (United Nations Environment Programme), 2014; Hamilton and Casey, 2016; Bryan-Brown et al., 2020; Goldberg et al., 2020]. Between 2000 and 2016, anthropogenic impacts were responsible for 62% of the global mangrove area loss, with shrimp, rice and oil palm cultivation responsible for close to half of these global losses (Goldberg et al., 2020). Coastal erosion (mainly because of sea level rise and alterations to river dynamics) contributed to 27% of global losses (2000-2016) and is the second largest cause of global mangrove loss (Thomas et al., 2017; Goldberg et al., 2020). Unfortunately, little attention has been given to the degradation of mangrove functions, such as nutrient cycling, species composition and biomass allocation, which directly affects ecosystems services, since large scale monitoring using remote sensing techniques is seldom applicable and field observation requires relatively long monitoring periods to cover the natural variability of environmental processes (Sanyal et al., 2020).

There is still a large gap in the knowledge concerning the geochemical and biogeochemical responses of mangroves to degradation drivers (Lourenço et al., 2020). Approaches considering the land-ocean continuum, as well as the transfer processes involved, are still scarce and mostly consist of short-term analyses. In addition, long-term studies covering large geographical scales are scarce and threatened by economic instability of science funding in several countries harboring mangroves in the world's tropical coasts (Lacerda et al., 2020). Of particular interest are studies that assess anthropogenic influences on the interaction between drainage basins and the continent-ocean interface, as well as impacts on ecosystem services, aimed at understanding the implications of global change on ecosystem functioning, conservation and sustainable development, the vulnerability of the continent-ocean interface, and threats to society through food security. The response of mangrove ecosystems to degradation drivers is frequently more intense and conspicuous in extreme environments, therefore, semiarid coastline are ideal sites for such studies.

Brazil has the third largest extent of mangroves extension in the world. Estimates of total area vary by 30%: 9,627 km^2 (Giri et al., 2011), 9,940 km^2 (Diniz et al., 2019), 10,123 km^2 (Food and Agriculture Organization [FAO], 2007), 13,626 km^2 (Lacerda, 2002), and 13,989 km^2 (Instituto Chico Mendes de Conservação da Biodiversidade [ICMBio], 2018). Bunting et al. (2018) suggested 11,072 km^2, as a more reliable figure based on detailed methodology and recent databases. The northeast coast bordering the semiarid hinterland spreads from about 2.7°S to 9.1°S and 41.5°W to 36.5°W, within the region "Northeast Brazil Large Marine Ecosystem," under a Bs (semiarid) climate. Details of the biology, geology, geomorphology and oceanography of this sector of the Brazilian coast can be found in Ekau and Knoppers (1999); Knoppers et al. (1999). The region harbors about 690 km^2 of mangroves, about 7% of the total Brazilian mangrove area (**Figure 1**), located in low-lying coastal plains that provide important environmental services for coastal populations (Instituto Chico Mendes de Conservação da Biodiversidade [ICMBio], 2018; Diniz et al., 2019). Due to the geographical location, semiarid climate, and resultant low terrestrial runoff to the continent-ocean interface, mangroves in this region are already under periodical environmental stress from natural drivers, including low annual rainfall, extended droughts and altered salinity (Marengo et al., 2018). As such, they are particularly sensitive to anthropogenic impacts from global (e.g., sea level rise; decreasing annual rainfall) (Godoy and Lacerda, 2015; Ward et al., 2016b; Ward and Lacerda, 2021) as well as regional drivers (e.g., damming; waste disposal) (Godoy et al., 2018; Loureiro and de Oliveira, 2019). For example, the combination of low continental runoff, river damming, increasing sea level and saline intrusion, results in mangroves in most world semiarid regions migrating landward (Ward and Lacerda, 2021).

Mangrove expansion inland has been observed in most estuaries along semiarid coastlines and is characteristic of the Northeast coast of Brazil and this represents the most important adaptation to climate change and regional environmental change, whilst requiring the existence of a large buffer zone along the mangrove-land border (Godoy and Lacerda, 2015; Ferreira and Lacerda, 2016). Therefore, anthropogenic activities in these areas, such as urban and tourism expansion, agriculture and aquaculture may pose strong constraints to mangrove adaptation and even their survival in this region.

Brazil established a Forest Code in 1965, which not only considered mangroves as fully protected areas, but also included broad expanses of salt flats, locally known as "*apicuns*," as integrated parts of the mangrove ecosystem. These flats stretch to the maximum influence of the tides or the extreme reaches of saline intrusion. In addition, coastal wetlands of the Northeast are exposed to long dry seasons and extended pluriannual drought (Marengo et al., 2018), facilitating the interpretation by developers and local authorities to acknowledge them as permanently dry habitats not considering the natural, broad temporal variation in water level as an inherent attribute of the system. This view, unfortunately, has led to a revision of the 1965 Forest Code following pressure from production sectors over the federal legislative chambers. This revision unlocked most of these flat plains to aquaculture and infrastructure development. Within the northeast region this "newly opened area" to development may reach over 6,000 km^2 (Ferreira and Lacerda, 2016). In addition, the review of the old Forest Code has also allowed forest conversion in alleged "public utility" or "social interest" projects, even in permanent protection areas, such as mangroves (Oliveira-Filho et al., 2016). Further setbacks to mangrove conservation were triggered by Brazilian Government recently (Bernardino et al., 2021) abolishing a wider range of the protective legal framework, though suspended by judicial order.

Because of the strengthening aridity, caused by a decrease in annual rainfall and extended drought periods linked to

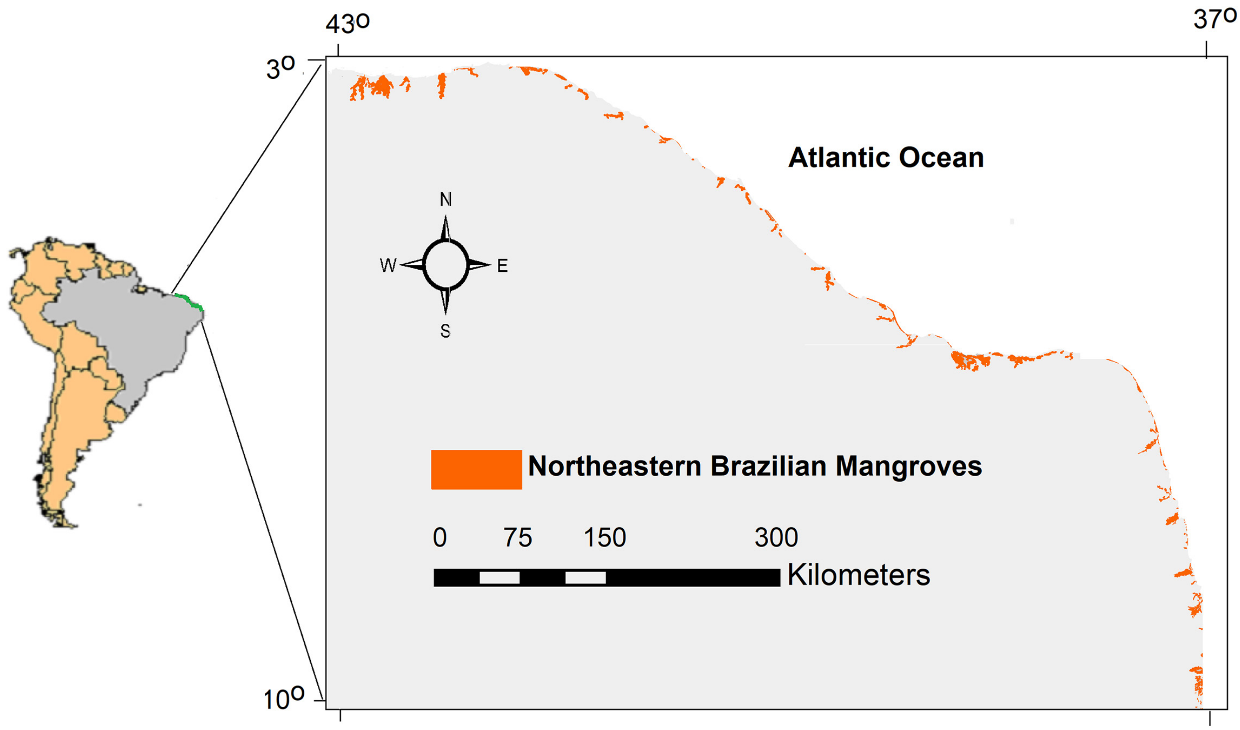

FIGURE 1 | Distribution of mangroves along the Northeast of Brazil. Mangroves of the semiarid coast, where most aquaculture is established, are shown in red. Map modified from Maia et al. (2006), Instituto Chico Mendes de Conservação da Biodiversidade [ICMBio] (2018).

climate change (Marengo et al., 2018), there is growing pressure on freshwater resources exacerbated by increasing human activity. This demand requires more dams and larger reservoirs, which have exacerbated the accelerated increase in sea level (Intergovernmental Panel on Climate Change [IPCC], 2019), and will result in a rapidly changing land-sea interface, where effects from human interventions are maximized. Therefore, conflicts between mangrove conservation and human occupation of salt flats are likely to escalate soon in this semiarid stretch of the northeast Brazilian coast, particularly related to shrimp farming, due to the accelerated rate of its development in the region. Also, there is a growing concern that these new exploitation areas can advance into the North Brazil mangroves, where around 70% of the country's mangroves are located, including the largest continuous mangrove strip on the planet. This raises a global concern to their conservation. Similar temporal trends, practices and outcomes of shrimp farming have been seen in countless mangroves around the world, so our findings can be considered globally to monitor estuarine waters and improve the conservation and management of mangrove forests.

This review article provides a synthesis of the impacts of aquaculture on ecosystem service provision and the environmental condition of associated mangrove habitats in Northeast Brazil, providing a review of existing literature, with emphasis on semiarid regions, through an expert based approach. Care was taken to avoid non-refereed publications, as well as general publications on environmental impacts on mangroves that are not supported by field data and observations. For the specific case in the Jaguaribe estuary, the largest single area of shrimp farming in Brazil, where over 3.34 km^2 of shrimp ponds has been built. We updated the remote sensing information to create a new map of the evolution of mangroves and shrimp farms in the area from 1992 to 2010, when aquaculture expansion was at its maximum, to show the little relationship between the increase of shrimp farms area and reduction in mangrove extension. The map is based on Landsat 5 images with 30-m spatial resolution. Images were obtained from the Brazilian National Institute for Space Research (INPE). Projection used was Universal Transverse Mercator (UTM), referenced to the horizontal geodetic datum SIRGAS 2000. Images were georeferenced using permanent reference points and the root mean square of geoprocessing error was less than 10m. Image vectorization was performed in ArcGIS 10 using a pixel-by-pixel supervised classification methodology. We hypothesize that rather than direct impacts on mangroves by deforestation and conversion, indirect impacts on ecosystem functioning are presently more significant, at least in semiarid climates.

SHRIMP FARMING EXPANSION IN NORTHEAST BRAZIL

The Pacific white shrimp *Litopenaeus vannamei* (Boone) is the most common species used in shrimp farming in the Latin America and Caribbean (LA&C) region. Although in the past century this activity was of minor environmental significance to mangroves throughout most of the continent, save for Ecuador, relative to other anthropogenic drivers, shrimp aquaculture has proliferated over the last three decades. From a few producer countries in 1990, to 22 out of 36 countries in the region were significant producers by 2017. The region's total annual production and pond area increased from approximately 86,000 t and 25,000 ha in 1990 to over 766,000 t and about 200,000 ha, respectively, by 2017 (Food and Agriculture Organization [FAO], 2019). This growth $\sim$ 20% per year has resulted in shrimp aquaculture being the major driver of environmental impacts on LA&C mangroves (Lacerda et al., 2019). Additionally, this globalized system of aquaculture production is energy-intensive, induces pressure upon local ecosystems and is, in general, highly dependent on marine capture fisheries for aquafeed production (Ahmed and Thompson, 2019). Therefore, instead of supporting sustainable development, shrimp aquaculture, in many situations, intensifies ecological degradation by focusing on the production of a high-value commodity d, based on production-intensive systems (Longo et al., 2013).

Over the past three decades, Brazil, as in the broader LA&C region, *L. vannamei* aquaculture has expanded from a few hectares in 1990 to nearly 20,000 ha in 2018. This is spread over several coastal states, but the Northeast region corresponds to 98% of the country's total shrimp production with 19,845 hectares of active ponds in 2018. Thus, the semiarid littoral region, reviewed here, corresponds to almost the whole of the country's shrimp pond area. Recent annual production statistics for the Northeast region reached 70,500 t in 2015, but decreased to 45,500 t in 2018 (Instituto Brasileiro de Geografia e Estatística [IBGE], 2018). This decrease was mostly caused by mortality because of viral diseases (Associação Brasileira dos Criadores de Camarão [ABCC], 2017; Carvalho and Martins, 2017). More recent estimates suggest a total area and production in 2020 of about 30,000 ha and 110,000 t (Freitas et al., 2017; Rocha, 2019), but there is high unreliability in those numbers.

In Northeast Brazil, during peak production years, export revenues reached US$ 240-270 million, making shrimp culture an extremely profitable activity, second only to sugar cane, surpassing traditional crops such as cashew nut and irrigated fruit agriculture (Costa and Sampaio, 2004; Sá et al., 2013). It has been claimed that the productive chain of intensive shrimp aquaculture can generate from 1.8 to 3.7 job ha^{-1}, like the job demands of the irrigated fruit agriculture sector (Costa and Sampaio, 2004), but it can frequently be lower (0.6 job ha^{-1}) (Monteiro et al., 2016). Farms in Rio Grande do Norte State have levels of 0.5 full-time job ha^{-1} to 1 job ha^{-1} during seasonal harvestings (*pers. comm.*). Indeed, in more than 75% of Brazilian farms (<10 ha in size) more than 41% of the employment is seasonal, and in extensive medium and large farms installed infrastructure requires fewer workers (Costa and Sampaio, 2004; Ministério da Pesca e da Agricultura [MPA], 2013). Whilst in some municipalities, aquaculture activity could contribute to municipal revenues, in general this contribution is mostly indirect and tax free (Sampaio et al., 2005, 2008). In several states, charged tariffs over the high water use of the activity are extremely low, including farms located in areas with chronic water shortages such as the semi-arid northeast coast (Monteiro et al., 2016). The contribution of shrimp farming to municipal GDP growth (Sampaio et al., 2005, 2008), does not necessarily mean direct employment nor development, since GDP growth is not a good indicator of development (Daly, 2005; Jackson, 2009). Some high productivity levels 30 t ha^{-1} y^{-1} (well above the national average of 2.56 t ha^{-1} y^{-1}, however) (Tahim and Araújo Junior, 2014; Oliveira and Neto, 2019; Rocha, 2019) could generate (private) income estimated at more than 120,000 US$ ha^{-1} y^{-1} (shrimp local values at 2019), whilst the cumulative environmental damage to air, soil, water, biota and landscape can reach around 4.2 and 4.6 US$ million ha^{-1} (Ferreira and Lacerda, 2016), mainly through the high carbon (C) footprint due to deforestation of mangroves (Belettini et al., 2018; Ferreira et al., 2019; Nóbrega et al., 2019). Potential C emissions from the conversion of mangroves to shrimp ponds in Northeast Brazil are extremely significant, approximately 413 $\pm$ 94 MgC ha^{-1} (Kauffman et al., 2018). However, for now the well-established technology and a constantly growing market forecast for the demand for shrimp, suggests there will be a continuity of aquaculture activity in Northeast Brazil.

IMPACTS ON MANGROVES FOREST COVERAGES

Globally, aquaculture is a significant driver of deforestation of mangroves (Friess et al., 2019). However, along the Northeast coast of Brazil, shrimp aquaculture is rather a driver of degradation, responsible for adversely affecting mangrove functioning through excess nutrient inputs (Sá et al., 2013; Marins et al., 2020), erosion of fringe mangrove forests (Godoy et al., 2018) and, to a lesser extent, illegal deforestation and conversion of mangroves to shrimp ponds (Ferreira and Lacerda, 2016).

Between 8% (Maia et al., 2006) and 10.5% (Instituto Chico Mendes de Conservação da Biodiversidade [ICMBio], 2018) of the total shrimp pond area is from mangrove deforestation. An unknown fraction of mangrove cover has been lost due to the opening of channels, changes in hydrological dynamics of coastal plains and sedimentation/erosion of tidal creeks and riverbanks. Since the Forest Code amendment requires no planning for decommissioning shrimp farms, many of the ponds deactivated in the northeast, due to disease and economic crises, are witnessing a new tide of transformation. Abandoned shrimp farms are being converted into cattle grazing areas, salt ponds and human occupation advances, rather than rehabilitating the original mangrove cover. This accelerates soil degradation and may eventually promote further expansion over the remnant mangroves (Nunes et al., 2011).

The semiarid northeast coast has less than 10% of the total mangrove area in Brazil (Instituto Chico Mendes de Conservação da Biodiversidade [ICMBio], 2018. Therefore, although only a small area of Northeast mangroves has been directly converted to aquaculture ponds (Maia et al., 2006; Instituto Chico Mendes de Conservação da Biodiversidade [ICMBio], 2018). This forest loss is proportionally more significant than in other mangrove regions, moreover, the coastal waters along the semiarid region are highly oligotrophic and most productivity depends on nutrient fluxes from the continent. Local artisanal fisheries, representing about 90% of the total catch of the region, are mostly based on mangrove associate species, such as crabs and oysters and other species dependent on mangroves for breeding, nursery and protection, mostly fish. Indeed, one hectare of preserved mangroves house around 5.1 t of mangrove crab *Ucides cordatus* and yield around 20 t of animal biomass per year including fishes, molluscs, and crustaceans (Instituto Brasileiro de Meio Ambiente [IBAMA] and Centro de Pesquisa e Conservação da Biodiversidade Marinha do Nordeste [CEPENE], 1994; Rocha Junior, 2011). This available food source makes a great difference for low-income families in the region, some of the poorest in the country. Conflicts between aquaculture and artisanal fishers have become frequent in the entire region.

A case study in the São Francisco estuary, based on local traditional fishers' perception on the causes of the observed local mangrove-based fisheries decline, showed that about 90% of the respondents suggested that the primary reason for the observed decline in catch was the increase in the number of fishers working in the area. The second biggest impact, highlighted by 60% of the respondents, was considered to be shrimp farming. Only 13% of the respondents suggested the increase in shrimp farms as a possible solution for the decline in artisanal fisheries (Santos et al., 2017).

The lower Jaguaribe River estuary has the largest concentration of aquaculture activities in Northeast Brazil. Between 1992 and 2010, the total shrimp pond area increased from 230 ha in 1992 to nearly 21,600 ha in 2010 (**Figure 2**). The original mangrove cover of 700 ha has decreased by 210 ha over the same period, but the total mangrove extent in this estuary increased by 240 ha over the same period due to expansion into former mud/sand flat areas as a result of changes in the watershed influenced by global and regional drivers (Godoy and Lacerda, 2014; Godoy et al., 2018). This difference in mangrove extent is not a result of direct conversion, as seen in the Potengi estuary (**Figure 3**), but because of the indirect death of trees due to hydrological changes cause by pond construction and the opening of channels for incoming and outgoing water. Mangrove expansion linked to climate change and river damming has in part counterbalanced this reduction in extent (Godoy and Lacerda, 2015; Lacerda et al., 2020).

In Northeast Brazil, the principal impacts from shrimp aquaculture on mangrove forest cover are indirect, as opposed to direct conversion as is more typical in other mangrove regions. A recent review of the environmental impacts associated with aquaculture in Mexico, predominantly also in a semiarid climate, highlighted the significance of indirect impacts, mostly from effluents altering environmental characteristics (Sosa-Villalobos et al., 2016). Hong et al. (2019) used multi-temporal Landsat data from a period of 30 years between 1988 and 2018 to reveal a decrease of 4,980 ha of dense mangrove forests (about 90% of the original vegetated area) and a decrease of 7,816 ha of sparse mangrove forests (about 55% of the original vegetated area), linked to an increase of 150,720 ha (5,024 ha year) of shrimp ponds in the SE Mekong River Delta, Vietnam. In Malaysia, about 11% of the dense mangrove forests and 21% increase of mudflats were lost between 1989 and 2017 along the Tumpat Kelantan littoral following the expansion of shrimp farms in the region (Rasid et al., 2019).

A global survey on the impacts of shrimp aquaculture on mangroves (Hamilton, 2013) showed direct mangrove deforestation in Brazil resulted in losses of about 13% of the total area, consistent to the figures discussed above, based on Maia et al. (2006), Instituto Chico Mendes de Conservação da Biodiversidade [ICMBio] (2018). In contrast, an average of 29.9% of Southeast Asian mangroves were converted to aquaculture ponds since 2001, with a high variation among the different countries (Murdiyarso et al., 2015; Richards and Friess, 2016). Area loss in Brazil is broadly comparable with Bangladesh (7%) and India (4%), but much lower than direct conversion rates observed in Vietnam (53%), Indonesia (48%), China and Ecuador (40%), and Thailand (19%). The time interval used in the research varied from country to country and was typically based on the first arrival of large-scale commercial aquaculture. The study showed a 51.9% loss of global mangrove area during the analyzed period, and estimated that commercial aquaculture accounted for 28% of the total mangrove loss (544,000 ha).

Altered Mangrove Functioning

Whilst not as evident as forest conversion, changes in mangrove functioning related to shrimp farming is more scarcely reported, particularly in semi-arid mangroves, where the shrimp farming is a strong driver of environmental changes, as well as regionally important economic activities (Queiroz et al., 2020). Major changes associated with shrimp farming relate to biomass allocation, forest architecture and health (Alatorre et al., 2016; Datta and Deb, 2017), species composition, cycling and mineralization of nutrients (Molnar et al., 2013; Nóbrega et al., 2013; Queiroz et al., 2020), accumulation of nutrients (Marins et al., 2020; Pérez et al., 2020; Queiroz et al., 2020), and C storages (Suárez-Abelenda et al., 2013; Ahmed et al., 2017; Nóbrega et al., 2019; Tian et al., 2019). The effect of shrimp farming on plant activity in mangroves has been investigated by direct measurements of ecological variables or indirectly through time series of the Normalized Difference Vegetation Index (NDVI). The basis of NDVI is the quantification of the amount of sun light reflectance at the canopy levels controlled by properties of pigments, water and C. Healthy vegetation absorbs sun light and reflects it back (Davaasuren and Meesters, 2012; Villoslada et al., 2020). Therefore, NDVI broadly compares photosynthetic activity and can be applied at spatial or temporal scales to monitor vegetation structure, phenology and biophysics (Wang et al., 2004), allowing monitoring of large stretches of mangroves.

Alatorre et al. (2016) showed the spatial relationship between the zones of greatest loss of mangroves and areas with a greater proliferation of shrimp farms in the Gulf of California, between 1990 and 2010. Over 30% percent of the total mangrove forest cover exhibited decreasing NDVI for the period associated with canopy degradation. Similarly, in the Northeast coast of Brazil, Marins et al. (2020) compared NDVI and P accumulation rates in two estuaries with different intensities of shrimp farming. A reduction of NDVI was obvious in the mangrove receiving effluents from the largest pond area. In addition, NDVI also decreases at the same pace as increasing pond area. A comparison

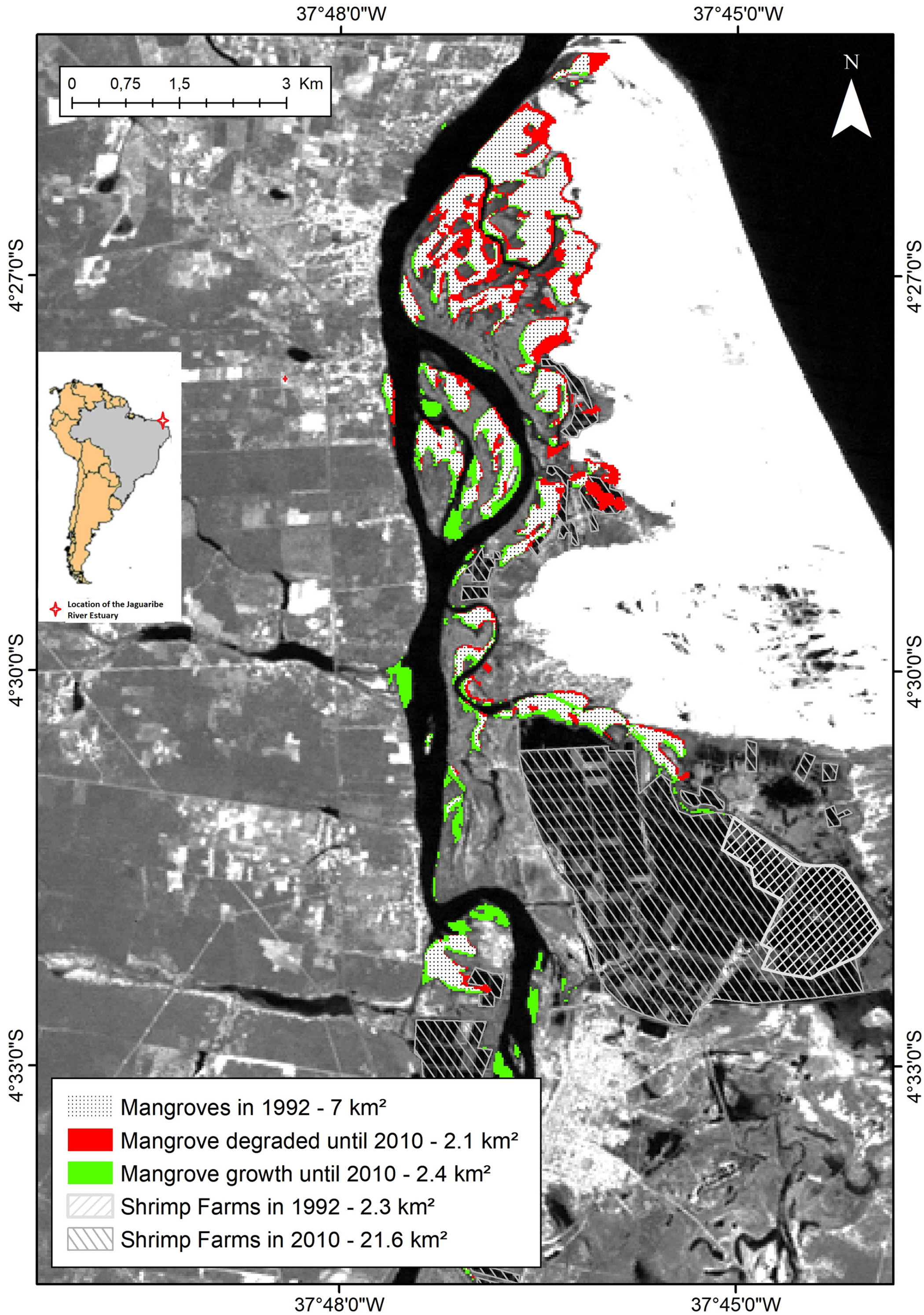

FIGURE 2 | Evolution of mangrove and shrimp farm areas in the Jaguaribe River estuary, the largest concentration of aquaculture activities in Brazil, between 1992 and 2010, when most expansion of the activity occurred.

of mangrove canopy structure in 2003 with 2017, in the Jaguaribe River, the location of Marins et al. (2020) study, is detailed in **Supplementary Figures 1**, **2**. There was a 15% reduction in NDVI from 0.78 in 2003, to 0.65 in 2008, following an increase of 340 ha to 1,600 ha of shrimp pond area. NDVI decreased further to 0.2 and lower in 2017, when the area of shrimp farms located

FIGURE 3 | Large mangrove cleared areas by shrimp farms at Potengi Estuary, Rio Grande do Norte State (Northeast Brazil). The mangrove island at the center-right of the image was almost totally converted to ponds.

in the forest drainage basin increase 10-fold to about 3,400 ha. It is interesting to note that there was no clear conversion of mangroves to shrimp ponds, but the occupation of tidal flats behind the mangroves has apparently affected hydrological processes, resulting in decreasing health of the mangrove fringe (**Supplementary Figure 1**). Unfortunately, this reduction in health is not computed as mangrove area loss reported in the literature, suggesting that more than 8% to 10% of mangrove forest loss relative to shrimp pond area, normally accepted as converted, is affected by hydrological changes promoted by pond installation in this region. Considering these results and the areas with NDVI < 0.1 (**Supplementary Figure 2**), literally equivalent to bare soil, between 9.6 and 14.4% of the local mangrove total were significantly affected by shrimp aquaculture.

Nutrient Dynamics and Eutrophication

Eutrophication, i.e., the exposure of coastal waters to excess nutrients, is a major anthropogenic phenomenon, and has been associated with the release of aquaculture effluents in various areas worldwide (Herbeck et al., 2013; Nóbrega et al., 2013). In Northeast Brazil, major anthropogenic drivers of coastal eutrophication are sewage in metropolitan areas, and agri/aquaculture, cattle husbandry and shrimp production, in rural areas. Since aquaculture presents much higher emission factors for major nutrients than other rural-area activities (**Table 1**) and effluents are directly emitted into estuaries or coastal areas (Lacerda et al., 2019), wherever significant pond areas exist, intense shrimp aquaculture effluents are the most important driver of eutrophication. Notwithstanding the high emission factors verified in Northeast Brazil shrimp farms, they are still at the lower range compared to other important producers using intensive cultivation processes. For example, N and P emissions from intensive shrimp aquaculture in Hainan, China is about 10-times higher than values reported for Northeast Brazil (31.9 and 1.3 t km^2 yr^{-1}, respectively), like other large Asian producers (Herbeck et al., 2013).

There is relatively little information on emission factors from aquaculture practices, thus the comparative figures between NE Brazil and China must be taken with care. However, in Latin America and the Caribbean, cultivated species and the technological packages are virtually the same. Also, as mentioned previously, most shrimp aquaculture in the region occurs under a semiarid climate. Therefore, the emission factors estimated for NE Brazil (**Table 1**) may be used to calculate emission loads at a regional level and may be significant to a worldwide evaluation of the relative contribution of shrimp farm effluents to coastal nutrients fluxes (Lacerda et al., 2006a,b).

The high reactivity of shrimp pond effluents, relative to agriculture and cattle husbandry facilitates rapid eutrophication.

TABLE 1 | Emission factors and total annual emission of nutrients (t km^{-2} year^{-1}) and metals (kg km^{-2} year^{-1}) from shrimp aquaculture in Northeast Brazil, after Lacerda et al. (2006a,b), Lacerda et al. (2011), Sá et al. (2013), León-Cañedo et al. (2017).

Element	Emission Factors from intensive shrimp farms	Emission Factors from Other non-industrial anthropogenic sources
Phosphorus	0.13 – 0.32	<0.01 – 1.73
Nitrogen	1.25 – 4.09	<0.01 – 2.65
Copper	38.6 – 59.8	<0.01 – 15.3
Mercury	0.03 – 0.04	<0.01 – 0.04

Other non-industrial anthropogenic sources include agriculture, animal husbandry, solid waste disposal, urban runoff, and sewage.

FIGURE 4 | Extensive algal bloom resulting from excess nutrients from shrimp farm effluents in the Jaguaribe River Estuary, Northeast Brazil.

For example, in the Jaguaribe Estuary, a major shrimp production area in Ceará State, anthropogenic P emissions increased by 30% to 43.9 t yr^{-1}, between 2001 and 2006, following the expansion of shrimp farms (Marins et al., 2011). Further expansion of local farms increased emissions to 69 t yr^{-1} in 2013 (Marins et al., 2020), representing over 60% of the total P emissions to the lower Jaguaribe Basin. Algal blooms are now frequently observed in this estuary (**Figure 4**).

Apart from triggering eutrophication, shrimp farm effluents also affect mangroves by decreasing the ecosystem's capacity to retain nutrients (Barcellos et al., 2019; Marins et al., 2020) and may increase mangrove C stock loss. Decreasing C accumulation capacity in mangrove soils is, in part, due to increasing organic matter decomposition by a bacterial community fueled by nutrient-rich aquaculture effluents and reducing pyritization due to increasing the concentration of strong electro acceptors, such as nitrates, which are also enriched in shrimp farm effluents (Suárez-Abelenda et al., 2013). Unfortunately, how this amount compares with losses from direct conversion of mangroves to shrimp ponds (58 - 82%) of the ecosystem C stocks (Kauffman et al., 2018), is still unaccountable.

A comparison of two estuaries in Brazil receiving effluents from urban sources and shrimp farms, with a pristine site, showed that shrimp farm effluents have stronger effects on mangroves than urban wastewater. The main cause for the difference in the degree of impact is the higher reactivity of P species from shrimp farm effluents, which are more enriched in exchangeable and soluble-P. Species of P in mangroves receiving wastewater are predominantly organic-P, oxide-P, and clay/Al-P (Barcellos et al., 2019).

Mangroves, like other tidal wetland vegetation, can decrease the impact of land-derived nutrient loads, by accumulation in soil and biomass (Ward et al., 2016a; Lima et al., 2020; Valiela et al., 2020). However, there is a critical level of nutrient load, meaning that there is a clear threshold to this filtering in mangroves (Valiela and Cole, 2002). In Northeast Brazil, an evaluation of the tidal exchange of suspended matter (TSS), total P (TP), soluble reactive P (SRP) and particulate P (Part-P), showed that the retention of P varies with the magnitude of shrimp farm effluents. In a site receiving effluents from nearly 3,000 ha of shrimp ponds, reaching from 1.2 to 5.2 kg hr^{-1} of total P, local mangroves could trap 40%, 45%, 47% and 70% of the TSS, TP, SRP, and Part-P, respectively, from the incoming high tidal flux. However, in another site, receiving P from only 10 ha of ponds (0.22 kg hr^{-1}), tidal balances showed a much higher relative retention of the total influx, over 92% of the total tidal input of TSS and all P fractions. These results suggest that mangrove P accumulation capacity is significantly decreased with increasing nutrient inputs and this limits the potential of mangroves to act as a natural barrier to nutrient transport (Marins et al., 2020).

Similar results were observed by Queiroz et al. (2020) relative to Nitrogen (N) dynamics in sediments from mangroves affected by shrimp pond effluents in the Jaguaribe Estuary, Ceará State. The tidal balance of N through creeks showed that only 30% of the mineralized N remains stored in the sediment, whereas 70% was solubilized in tidal waters. Therefore, N mineralization may induce eutrophication by augmenting inorganic N bioavailability in mangroves receiving N-rich effluents from aquaculture, triggering increases in primary productivity.

Elemental ratios of nutrients in mangrove sediments were also highly affected by shrimp pond effluents. Total N content (13%) and C:N ratio (9.6) were much higher in sediments downstream of shrimp ponds than those from upstream sediments (TN = 3%, C:N = 4.2). Simultaneous analysis of aquafeeds used in the local shrimp farms also showed a high C:N ratio (8.0) and total N content (5%) (Zocatelli et al., 2007). Phosphorus (P) distribution is also affected by effluents. Concentrations of dissolved-P (3.1 μM and 5.6 μM) and particulate-P (2.1 – 6.5 μM and 1.3 – 11.9 μM) in mangrove tidal creeks receiving shrimp farm effluents are much higher than concentrations observed in mangrove creek waters not receiving aquaculture effluents, where average concentrations of dissolved-P (0.5 μM) and particulate-P (4.1 μM) were 8 to 1.5 times lower, respectively (Marins et al., 2020).

Similar results showing increasing nutrient content and changing elemental ratios related to shrimp farming have been reported in other areas worldwide. In *Kandelia obovata* forest sediments, along Jiulong River Estuary, in Fujian, China, shrimp pond effluents from 8-year-old farms were shown to have increased soil TOC and TP contributing to 30.0 to 33.6% of the total TOC within mangrove surface sediments (0 – 10 cm depth) (Tian et al., 2019).

Impacts of C losses from mangroves associated with conversion to shrimp aquaculture include the removal of above- and below-ground biomass, loss of soil organic C, and decreases in retention and sequestration through alterations to the hydrological regime and retention of allochthonous C as a result of both this, and removal of the vegetation canopy (e.g., periodically inundated roots and associated epiphytes). In fact, Arifanti et al. (2019) have suggested that conversion to aquaculture could result in C losses equivalent to 226 years' worth of accumulation. It is not fully clear how mangroves will respond to restoration measures following aquaculture abandonment. Some studies have shown that there is a rapid increase in soil organic matter accumulation following mangrove restoration (Lunstrum and Chen, 2014; Osland et al., 2020) as well as above- and below-ground biomass (Charles et al., 2020). However, this will depend on the hydrological regime, soil surface micro-climate, inorganic and organic pollution levels, the relative spatiotemporal scales of mangrove vegetation change, sodicity, and sources of organic matter (whether recalcitrant or labile) (Suárez-Abelenda et al., 2013; Tran et al., 2015; Celis-Hernandez et al., 2020; Charles et al., 2020).

Reforested mangroves in arid climate zones typically sequester C in biomass 10 times slower than equivalent sites in high rainfall tropical zones (Sasmito et al., 2019). However, considering the generally low biomass of coastal zone vegetation in semiarid climates, frequently restricted to mangroves, the relatively lower C accumulation may still be significant at a regional level. As highlighted for many different regions such as the Gulf Coast (Sheppard et al., 2010; Saderne et al., 2018); East Africa (Benson et al., 2017); the Pacific (Alatorre et al., 2016); and the Caribbean coasts of Mexico (Adame et al., 2013).

The greatest impact from aquaculture on C stocks in mangroves is through organic soil losses from upper horizons to form ponds, where the bulk of C is stored (e.g., the top 1 m organic rich soil layer) (Kauffman et al., 2018). Kauffman et al. (2018) estimated that 67% of the soil organic C is stored in the upper 1m in mangroves in the semi-arid Northeast of Brazil and that C losses from the soil are likely to account for 81% of the C lost following conversion to aquaculture. Whilst shrimp ponds do start to store C in their soils following conversion, this is substantially less than adjacent mangrove areas due to decreases in C density in the soils, as well as reductions in accumulation rates (Eid et al., 2019). Aquaculture conversion contributes to higher C loss (72 ± 44 Mg C ha^{-1} or 83% ± 37%) in above and below biomass of mangroves, whereas abandoned ponds still release soil organic C and GHG continuously via both soil-water and soil-air interfaces. Regeneration of biomass levels in general last around 40 years, but soil levels of C take longer to return (Sasmito et al., 2019). It has been suggested that mangroves can lose up to 70% of their C through conversion to aquaculture, although where appropriate hydrological connectivity is maintained natural mangrove rehabilitation can occur, together with associated recovery of carbon stocks over time (Matsui et al., 2012; Elwin et al., 2019). Nam et al. (2016) suggest that, where hydrological connectivity is maintained, there is unlikely to be a substantial difference in carbon accumulation or stocks between planted and natural rehabilitation sites, which is important considering the increasing acknowledgment of this vital ecosystem service for climate change mitigation. However, where hydrological connectivity is not restored appropriately, it is likely that shrimp ponds will convert to unvegetated mud/sandflats, which typically contain less soil organic carbon and limited associated biomass than mangroves (Lunstrum and Chen, 2014).

Trace Metals Derived From Shrimp Farming

The high productivity of ponds depends on large amounts of aquafeeds and fertilizers to induce production, and strong aeration of ponds to avoid oxygen depletion. As a result, effluents are enriched in nutrients and organic matter from excess fertilizers, aquafeeds, ecdysis, and suspended matter due to erosion of pond walls by the aerators (Lacerda et al., 2006a). Due to the large amounts of aquafeeds used, impurities present in them as well as in other materials (e.g., fertilizers, lime, and chloride), such as trace metals, may also accumulate within the pond environment and thus be present in the effluents (Boyd and Massaut, 1999; Chou et al., 2002; Lacerda et al., 2009, 2011). Some trace elements, such as Cu and Hg, whose emission factors are particularly high relative to other anthropogenic sources (**Table 1**), are of environmental significance and represent an additional environmental threat. Another trace element of environmental significance is Zn, which has also shown to be affected by shrimp pond effluents (Silva et al., 2001) and seems to present high emission factors (León-Cañedo et al., 2017). Unfortunately, the one study, to our knowledge, that has analyzed Zn in Northeast Brazil mangroves observed an increase in Zn concentrations in oysters and sediments downstream of shrimp farm effluent outfalls (Silva et al., 2001).

Table 2 summarizes Hg and Cu concentrations in aquafeed and other products used in intensive shrimp aquaculture in NE Brazil, compared to concentrations found in sediments and suspended particles of the environment surrounding the farms. Shrimp farm emission factors (**Table 1**) result in annual emissions of 490 kgCu yr^{-1} and 0.35 kgHg yr^{-1} to Northeast Brazilian

TABLE 2 | Concentrations of Hg and Cu in aquafeeds and chemicals used in intensive shrimp farms in Northeast Brazil and environmental levels in local mangroves, from Lacerda et al. (2006a), Lacerda et al. (2011), Costa et al. (2013).

Material	Cu (μg g^{-1})	Hg (ng g^{-1})
Aquafeeds	34 – 52	24 ± 18
Fertilizers	0.7 – 1.9	4.9 ± 4.9
Lime	1.9 – 3.2	17 ± 5
Outfall total suspended particles (TSS)	2.1 – 68.2	9 – 179
Mangrove TSS upstream farm's effluent:	34	28.1 - 53.2
Mangrove TSS downstream farm's effluent:	542	55.8 – 146
Pond sediments	9.7 – 15.2	-
Creek sediments upstream farm's effluent:	1.4	5.1 - 6.6
Creek sediments downstream farm's effluent:	10.1	1.6 - 10.3

estuaries harboring shrimp farms. Aquafeeds and fertilizer are the principal sources of the metals to the aquaculture process. Notwithstanding, concentrations of Cu and Hg in farmed shrimp are low (40 μgCu g^{-1} and 17 ngHg g^{-1}), posing no toxicological threat to human consumption. However, accumulation of both metals in exoskeleton compared to muscle tissue, suggest that detoxifying mechanisms are taking place and can thus impact shrimp growth and the economic efficiency of farms (Lacerda et al., 2006a, 2009; Soares et al., 2011).

Cu and Hg distributions in surface sediments from a tidal creek receiving shrimp farm effluents confirm the importance of this source at the regional level and vertical distribution of metal concentrations in sediment cores suggest a recent increase contemporaneous with aquaculture development in the region. In a comparison between concentrations of Cu and Hg from similar areas at the Northeast coast lacking intensive shrimp culture, concentrations of Cu and Hg were up to five times higher in sediments and biota from estuaries with shrimp farms. The results summarized in **Table 2**, clearly show the effects of shrimp farm effluents on increasing environmental concentrations of Cu and Hg in deposited and suspended sediments.

Costa et al. (2013) observed an increase in Hg concentrations in sediment cores within a mangrove creek receiving shrimp farm effluents at the Jaguaribe River estuary in Northeast Brazil. Levels increased from background concentrations ($<$0.5 – 3.6 ng g^{-1}) prior to the onset of shrimp farming to 7.2 – 11.7 ng g^{-1}, in present day sediment layers. Concentrations in muscle tissues of mangrove catfish were also higher in the creek sector closer to shrimp farm effluents relative to those found further away in the main river course. In Todos os Santos Bay, Bahia State, Northeast Brazil, Hatje et al. (2019) suggested that upstream shrimp farms were a point source of Hg and possibly other trace metals to their downstream mangrove study site. This resulted in a clear increase in total Hg concentrations in both upstream and downstream sites closest to the effluent outfalls. They noted, however, that the spatial distribution of Hg concentrations implies an impact over a relatively small area.

Contrary to most sources of contaminants (agriculture, animal husbandry and solid waste disposal from urban areas) present in this sector of the Brazilian coast, effluents from shrimp farming are not released onto soils prior to reaching the estuary. Due to the high emission factors and direct input to the estuarine system, shrimp aquaculture is the major source of nutrients and metals in most rural areas of the Northeast coast.

Unfortunately, the few studies on the effects of trace metals on mangroves are based on observations in severe industrial and urban polluted mangroves, that receive very large loads of such contaminants. Most are performed under controlled conditions, to study the effects of trace metals on mangrove plants in detail. Most suggest a decrease in photosynthesis, growth, and biomass (Maiti and Chowdhury, 2013). Although field observation reports on the low capacity of mangrove plants to absorb most metals, the same may not be true for mangrove animals (Thành-Nho et al., 2019). Notwithstanding, a key issue regarding metal accumulation and eventual harmful effects on mangrove biota depend on metal abundance and availability (Marchand et al., 2011; Lacerda et al., 2020). Unfortunately, no study to our knowledge has observed the effects of aquaculture derived trace metals on mangroves and this is an urgent necessity in view of the rapid expansion of the activity and the high metal-complexing capacity of dissolved C present in shrimp farming effluents (Lacerda et al., 2006a,b, 2011; Hidayati et al., 2020).

MANGROVE SERVICES AND SOCIOECONOMIC IMPACTS OF SHRIMP FARMING

Mangroves goods and services provided to human populations are manifold: coastal protection (Marois and Mitsch, 2015; Veetil et al., 2019), estuarine filtration (Celis-Hernandez et al., 2020), local climate regulation (Crona, 2006; Neogi et al., 2016), C sequestration and storage (Lee et al., 2014; Mafi-Gholami et al., 2018), fisheries (Aburto-Oropeza et al., 2008; Hutchison et al., 2014; Carrasquilla-Henao and Juanes, 2017), habitat for biodiversity (Nagelkerken et al., 2008; Barbier et al., 2011), and cultural values to local communities (Queiroz et al., 2017). In Northeast Brazil, the importance of these services is heightened by pressing socioeconomic hardship faced by a large part of the population, including poverty and hunger (Ottonelli and Mariano, 2014; Caldas and Sampaio, 2015) and precarious employment (da Silva Filho and de Queiroz, 2011). Added to these issues is a history of environmental degradation that dates back to the early decades after the Portuguese invasion (Machado, 2008).

Initial settlement by the Portuguese in Northeast Brazil was focused on the coastal areas of the region, where mangroves are located. Countrywide, this is still reflected today: a quarter of Brazil's population live on the coast. Alongside numerous large-scale enterprises on the Northeast coast, such as high-environmental-impact ports (Koening et al., 2002; Ferreira et al., 2012), and tourist resorts (Cardoso, 2005; Sousa et al., 2016), the Northeast coast is populated by fishing communities that are highly dependent on the goods and services locally provided by mangroves, especially food from fisheries (Vasconcellos et al., 2007). Per year, one hectare of preserved mangroves can yield 20 tons of animal biomass (Rocha Junior, 2011) and generate around US$ 40,000 of economic value from fisheries (Aburto-Oropeza et al., 2008).

Mangroves, as nurseries for fisheries resources, directly or indirectly support more than 1 million people in Brazil (Prates et al., 2012). When analyzed in connection with the ecosystems where they live and obtain their livelihoods, these fishing communities form social-ecological systems. This intrinsic dependence of local populations on mangroves makes them particularly vulnerable whenever mangrove goods and services are threatened or reduced by loss or damage to the mangroves.

Since shrimp farming is largely responsible for the degradation of mangroves in Northeast Brazil, consequently it is also responsible for the loss of related ecosystem goods and services that constitute the livelihoods of vulnerable populations that live on the Northeast coast. Some benefits can be locally provided by this activity, e.g., an increase and stability of employment and of income, leading to increased municipal revenue and improved living conditions in Northeast Brazil (Sampaio et al., 2008). Conversely, another study showed that only a very low percentage of the local population work in shrimp farms in the region (Safadi, 2018). It is likely that the few benefits are fully outweighed by the environmental impacts detailed in this review. And although some shrimp farms implement measures to lessen environmental impacts, efforts are mostly concentrated on increasing technical production efficiency when compared to those that aim to improve environmental quality (Araújo et al., 2018).

Queiroz et al. (2013) associated the rapid growth of shrimp farming in Ceará, Northeast Brazil, with an environmental and socioeconomic degradation of the mangrove system, which reduces the availability of services and compromises the socioenvironmental sustainability for the medium-and long-term. Impacts include increases in poverty, lack of land, food insecurity, displacement of local communities, water contamination, and poor working conditions. In Bahia state, for example, the lack of public protection policies has allowed the establishment of shrimp farms without sustainable productive alternatives that consider employment and food production (Dias et al., 2012). The perception of fishers regarding shrimp farming show that local communities do not necessarily enjoy the possible benefits from shrimp farming and are impacted by this activity due to the devastation of mangroves, impediment of passage, damage to fishing, among others. In traditional fishing communities in Northeast Brazil, typically only 20% of the local population worked on shrimp farms, yet the whole community is impacted by the activities (Safadi, 2018).

Alternatives to traditional shrimp farming practices have been shown to provide positive social outcomes. In Asian countries, mixed mangrove shrimp systems are sometimes used, where ponds are located in the ditches between the rows of mangrove trees that have been planted on platforms where water is exchanged only when needed for the management of the shrimp and other aquatic organisms (Bosma et al., 2014). These have a lower capital requirement compared to other shrimp farming systems, livelihood diversification through polyculture, provision of regular income and recognition as an organic farming practice (Bosma et al., 2014).

While it has been suggested that proper planning and management and alternative practices could help lessen the negative impacts of shrimp farming, it is essential to highlight that, like many other socioeconomic activities performed under the current capitalist mode of production, there is no evidence that it is possible to produce shrimp that can be sold in domestic or international markets in a competitive and upscalable way and, at the same time, generate socioeconomic benefits that clearly compensate for the social and environmental externalities generated by the activity. To quote Queiroz et al. (2013): "Shrimp aquaculture obeys the logic of appropriation of space generating socio-environmental consequences and compromising the flux of ecosystem services produced by mangroves."

The social-environmental impacts of shrimp farming have also been detected in other mangroves worldwide. These include the privatization of water and common-use public lands, the expulsion of ancestral fishing and indigenous populations, deforestation of mangroves, water contamination, depletion of fish stocks, salinization of aquifers, and loss of biodiversity, impacting food security and subsistence for mangrove populations (Polidoro et al., 2010).

At the local level, and in close collaboration with fishing and other extractive communities, the implementation of protected areas seems to be a key conservation strategy, especially by preventing or strongly regulating the installation of shrimp ponds. Restoration projects, as explained above, could potentially help lessen the negative impacts generated by shrimp farming. Local legislation could also help hinder degradational shrimp farming practices, but, in Brazil, these are subjected to federal regulation, which, as we already mentioned, was recently changed and, since those changes, exposes "*apicum*" areas to even greater threats (Schaeffer-Novelli et al., 2012; Ferreira and Lacerda, 2016; Borges et al., 2017).

Restoration of Deactivated Ponds

An estimated 1.4 million hectares of mangroves were lost in the world due to shrimp culture at the beginning of the 21st century (Valiela et al., 2020). By 2010, Aide et al. (2013) estimated a loss of 91.400 hectares of mangroves in LA&C alone, from which around 54,600 hectares were lost to shrimp aquaculture (Ahmed et al., 2018). Conversion of mangroves to aquaculture ponds and subsequent restoration has been undertaken in many countries including Sri Lanka, Thailand, Philippines, Indonesia, Brazil (Stevenson et al., 1999; López-Portillo et al., 2017) and other countries with unpublished data, like Ecuador and some Caribbean nations. Mangroves are resilient ecosystems that can self-repair in natural conditions. Depending on the level of degradation, abandoned shrimp ponds, if opened to input of tides and estuarine waters, can self-recuperate in 15 to 30 years and this seems to be the first and most effective solution to commence restoration (Stevenson et al., 1999; Matsui et al., 2010; Primavera et al., 2011; López-Portillo et al., 2017). Eliminating impairing or stressing factors e.g., dams or altered hydrology allows the influx of estuarine water to the ponds, and can supply areas with waterborne propagules, enabling recovery.

However, the effectiveness of this strategy and the time necessary to observe successful restoration will depend on the magnitude of the alterations in soil physicochemical features, such as acidification by sulfate oxidation (due to soil intervention by pond construction), desiccation, erosion or hardening/compaction, as well as hydrology (Stevenson et al., 1999; Di Nitto et al., 2013). Soil elevation and the lack of hydrological connectivity are factors that impair mangrove recovery (Wodehouse and Rayment, 2019). Leaving dams and enclosures of ponds after shrimp and salt work enterprise abandonment is a common problem that impairs the deposition of waterborne propagules and the repair of soil features, delaying mangrove recovery, sometimes by decades (Reis-Neto et al., 2019).

Artificial restoration is often necessary when the degradation of edaphic or hydrological conditions impairs the settling and development of propagules, when the homeostasis of the system has been disrupted, or where there is a need for rapid restoration (Lewis and Streever, 2000; Lewis, 2005; Ferreira et al., 2015). The best management practice is to restore a varied set of native species present before the degradation. Matsui et al. (2010) opened the banks of an abandoned shrimp pond and planted four native mangroves, among them *Rhizophora apiculata* and *R. mucronata*, and in six years, another fifteen mangrove species colonized the area. However, in sites where monospecific forests were cleared, or when only one species can cope with the soil/hydrological conditions, the planting of such specific species can be a shortcut to stand development, allowing the conditions necessary for the development of other mangrove species. *Avicennia germinans* has already been used for restoration in hypersaline soils in degraded mangrove areas in semi-arid estuaries (Toledo et al., 2001; Flores-Verdugo et al., 2015). *Rhizophora* spp. have been used extensively for their ecological tolerance, survival rate, rapid growth and easy planting, and these species can rapidly develop and restore some mangrove functionality, while other trees usually colonize later (Ferreira et al., 2015; López-Portillo et al., 2017; Eddiwan, 2018). Where many mangrove species are required, it is preferable to plant several species during initial restoration activities (Primavera et al., 2011).

Sometimes restoration of shrimp culture damage is costly, and in Brazil, offenders are not obliged to pay it. Brazil, like several other developing countries, suffers from a lack of political will to enforce environmental laws and increase monitoring of mangroves. Mangrove restoration of former shrimp ponds faces several socio-political difficulties. Community engagement/co-management arrangements can help improve organization and raise awareness and create a sense of responsibility over natural resources that are crucial to restoration success (Primavera et al., 2011; Ferreira and Lacerda, 2016; Borges et al., 2017; Hai et al., 2020). After the destruction of more than 75% of their mangroves, the Philippines initiated a special committee to integrate and rationalize programs of mangrove conservation, protection and restoration (Primavera and Esteban, 2008), and this could be a measure to formulate specific policies and protect mangroves in other countries, including Brazil.

In the Northeast of Brazil, several pond areas remain abandoned. Sea level rise is starting to enable the recovery of mangroves, as we observe in an experimental restoration in an abandoned saltpan constructed over an "*apicum*" (or salt flats) in the Pacoti estuary (Ceará State). More than three decades after deactivation, the saltpans have been partially colonized by *A. germinans*, because of increased sea level promoting overtopping the dikes that enclose the area. In a test to assess the influence of hydrological restoration on the planted *Rhizophora mangle*, increased influx of saline water resulted in high survival rates as well as an influx of natural seedlings of *Laguncularia racemosa* and *A. germinans*, which established in two years (*pers. comm.*). This showed that these abandoned ponds, and several similar level "*apicum*" areas, are subjected to natural (and sometimes rapid) regeneration if hydrological connectivity is restored. This suggests that these "*apicum*" wetlands are really part of the mangrove ecosystem and offer suitable mangrove refuges in the face of sea level rise. This also adds to the increased concern regarding recent Brazilian legislation, which puts at risk thousands of "*apicum*" areas by allowing their conversion to "productive" spaces such as shrimp ponds (Ferreira and Lacerda, 2016; Borges et al., 2017).

Mangrove restoration can also be used to return a range of ecosystem services including C sequestration and storage and sustainable fish, crabs or shrimp breeding (Fitzgerald, 2000; Lewis and Gilmore, 2007; Bosma et al., 2014; Ahmed et al., 2018; Ferreira et al., 2019). Shrimp production as practiced presently in Brazil shows the consequences of privatizing public environmental patrimony (and profit) while socializing environmental damage, because the abandonment of ponds leaves areas that formerly provided a range of goods and services as brownfield waste sites.

UPSCALING THE RESULTS

In Northeast Brazil, direct conversion of mangroves into shrimp farming ponds is less significant than in other LA&C and Asian countries. However, indirect effects on mangrove functioning, as reported by decreasing canopy health through NDVI estimates based on satellite images, reduction of sediment and nutrient accumulation capacity, as well as increasing GHG emissions from affected areas occur wherever adjacent mangrove areas are occupied by shrimp aquaculture. This makes it difficult to use experience from other major shrimp producer countries to forecast scenarios of the impacts from the activity on mangroves in Northeast Brazil.

Emission factors of nutrients and some trace metals of environmental significance are, in some cases, orders of magnitude higher from intensive shrimp aquaculture than other anthropogenic activities. This and the direct emission of effluents in mangrove creeks and estuaries, has been associated with eutrophication and contamination of sediments and biota in several world regions. Unfortunately, notwithstanding their significance for environmental health, trace metal contamination from shrimp farm effluents has received only minor attention from researchers, and no regulation from environmental authorities.

Since northeastern mangroves represents only 10% of Brazil's total area of these forests, conservation of existing forests and restoration of degraded mangroves and associated goods and services, is an urgent task both for the environment and local human populations. Restoration of mangroves requires systematic planning, societal awareness and commitment, public policies to shelter, coordinate and undertake mangrove restoration/rehabilitation, together with robust scientific knowledge to support successful schemes.

The recent changes in Brazilian forestry legislation, which resulted in opening "*apicum*" areas adjacent to mangroves for aquaculture development, allow for an intensification of environmental impacts to mangroves, and a real and present threat to the survival of traditional local populations that are still highly dependent on mangrove goods and services in Northeast Brazil. To fully understand this threat, however, future research should be based on valuation surveys and participatory methods, designed and consistent with international studies, such as focus groups and participant observation, in order to gather information and actively involve traditional local communities.

Finally, the major observations in this review apply not only to the NE Brazil region, although this sector of the Brazilian coast is responsible for the near totality of shrimp aquaculture in the country, but to other similar environmental settings. Major producers (Ecuador, Brazil, and Mexico) of aquaculture shrimp in Latin America and the Caribbean, use the same species (*L. vannamei*) and farms are preferentially located in semiarid coasts; therefore observations are likely to apply at a regional scale. In addition, other degradation drivers on mangroves, such as global warming, agriculture runoff and changing hydrology, will trigger similar environmental impacts as those from shrimp aquaculture reviewed here. Supporting integrated multinational initiatives to study indirect effects of aquaculture, rather than solely the effects on mangrove extension loss and gains should be a priority.

AUTHOR CONTRIBUTIONS

LL, RW, and AF conceived and wrote the manuscript. MG, RB, and AA contributed respectively to the remote sensing mapping and the analysis of human drivers, impacts and response, and had participated in the writing and reviewing the original manuscript. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/ffgc.2021.653096/full#supplementary-material

Supplementary Figure 1 | Impact on mangrove forest canopy health by increasing adjacent shrimp farm area. Note that direct conversion of mangroves into ponds is relatively small, but degradation of forest cover is widespread. Jaguaribe River estuary, Ceará State, NE Brazil.

Supplementary Figure 2 | Impact on mangrove forest canopy health by increasing adjacent shrimp farm area. Note that degradation areas present NDVI similar to bare soil. Jaguaribe River estuary, Ceará State, NE Brazil. Data from 2003 and 2008 from Marins et al. (2020); and for 2017, estimated in this study.

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Structural Characteristics of the Tallest Mangrove Forests of the American Continent: A Comparison of Ground-Based, Drone and Radar Measurements

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The Panama Bight eco-region along the Pacific coast of central and South America is considered to have one of the best-preserved mangrove ecosystems in the American continent. The regional climate, with rainfall easily reaching 5–8 m every year and weak wind conditions, contribute to the exceptionally tall mangroves along the southern Colombian and northern Ecuadorian Pacific coasts (Nariño Department and Esmeraldas Province areas). Here we evaluate the use of different methods (ground-based measurements, drone imagery and radar data [Shuttle Radar Topography mission-SRTM and TanDEM-X]) to characterize the structure of the tallest of these forests. In November 2019, three mangrove sites with canopy heights between 50 and 60 m, previously identified with SRTM data, were sampled close to the town of Guapi, Colombia. In addition to *in situ* field measurements of trees, we conducted airborne drone surveys in order to generate georeferenced orthomosaics and digital surface models (DSMs). We found that the extensive mangrove forests in this area of the Colombian Pacific are almost entirely composed of *Rhizophora* spp. trees. The tallest mangrove tree measured in the three plots was 57 m. With ca. 900 drone photographs, three orthomosaics (2 cm pixel^{-1} resolution) and digital surface models (3.5 cm pixel^{-1}) with average area of 4,0 ha were generated. The field-measured canopy heights were used to validate the drone-derived and radar-derived data, confirming these mangrove forests as the tallest in the Americas. The drone-derived orthomosaics showed significant patches of the Golden Leather Fern, *Acrostichum aureum*, an opportunistic species that can be associated to mangrove degradation, indicating that the mangrove forests investigated here may be threatened from increased selective logging requiring improvements and effective implementation of the current mangrove management plans in Colombia. The techniques used here are highly complementary and may represent the three tiers for carbon reporting, whereby the drone-derived canopy height maps, calibrated with local *in situ* measurements, provides cheap but reliable Tier 3 estimates of carbon stocks at the project level.

Keywords: unoccupied aerial vehicles (UAVs), Shuttle Radar Topography mission, mangrove above-ground biomass, eastern Pacific, Colombia, mangrove degradation, structure from motion (SFM), *Rhizophora* spp.

INTRODUCTION

Spaceborne remote sensing (SRS) is nowadays a key tool to understand Earth phenomena including the monitoring of biodiversity on our planet (Pettorelli et al., 2016). This is the case for the use of SRS in monitoring mangrove ecosystems. Accelerating since 2000 and facilitated by the free availability of satellite products, different global datasets of mangrove cover and loss are now available (Worthington et al., 2020). These baseline products (e.g., Giri et al., 2011; Bunting et al., 2018) have helped to derive global datasets of important properties of mangrove such as above-ground biomass (AGB) and carbon (AGC) after extrapolating or averaging site-specific values (Rovai et al., 2016; Rovai et al., 2018; Hamilton and Friess, 2018). Other SRS products like Lidar and interferometric synthetic aperture radar (InSAR) have been recently used to estimate mangrove heights at national, regional and global scales greatly improving previous AGB estimates (Fatoyinbo and Simard, 2013; Shapiro et al., 2015; Lagomasino et al., 2019; Simard et al., 2019). Nevertheless, there are still limitations faced when using some of these SRS global products with respect to their resolution (usually > 25 m per pixel) and the climatic conditions under which images can be taken (e.g., cloud free conditions).

Ideally, local and national mangrove mapping datasets that use remote sensing products with higher resolutions and that incorporate ground-truth measurements can play an important role in providing training data and validation for many of the on-going global mapping exercises (e.g., Global Mangrove Watch initiative; Worthington et al., 2020). Hence coordination and integration of global, national, and local datasets are needed to overcome many of the limitations of individual datasets.

Consumer-grade drones, belonging to the unoccupied aircraft systems (UAS) sector, appear to be an appropriate tool to overcome some of the limitations of SRS products in mangrove areas that are < 100 ha (Castellanos-Galindo et al., 2019). Examples of mapping mangrove forest with drones at spatial resolutions with centimeter scale are rapidly appearing in the literature (Otero et al., 2018; Li et al., 2019; Navarro et al., 2020). These finer resolution products provide unique opportunities to understand ecological processes and derive essential biodiversity variables that would not be possible to obtain with SRS at the moment. The use of UAS in mangrove mapping is therefore likely to simplify and complement traditional field inventories and additionally increase the accuracy of AGB and AGC estimates at the local level (e.g., Lucas et al., 2020). Collecting drone-derived information can therefore greatly benefit the calibration of the regional and global estimates that have been produced so far using SRS products of coarser resolution.

The Panama Bight eco-region in the west coast of Central and South America is considered to have one of the best-preserved mangrove ecosystems in the American continent and is thus a widely recognized conservation hotspot (Spalding et al., 2007). Highly developed mangrove forests are found along the southern Colombian and northern Ecuadorian Pacific coasts (Nariño and Esmeraldas areas; Castellanos-Galindo et al., 2015; Hamilton et al., 2017). Already in the mid-twentieth century, the American Geographer Robert West referred to these intertidal forests as the "most luxuriant mangroves of the world" (West, 1956). A more recent study (Simard et al., 2019) has identified the southern Colombian Pacific coast (Cauca and Nariño Departments), together with Gabon and Equatorial Guinea in Africa, as the regions containing the tallest mangroves in the world. A combination of high precipitation (i.e., potentially lower salinity), high temperature and low cyclone landfall frequency plus local geomorphological factors are associated to the observed exceptionally large areas and high canopy heights observed in those regions (Simard et al., 2019).

Mangroves in the southern Colombian Pacific coast occur in areas with geomorphological features like alluvial plains that form dynamic barrier islands surrounded by tidal channels (Martínez et al., 1995). These mangroves may be the wettest in the world with rainfall easily reaching 5 to ca. 8 m every year. The remoteness of this vast area (with no coastal road and only accessible by long boat journeys or small airplanes) contributes to the relatively pristine nature of some of these forests, but also to the difficulty in obtaining scientific ecological information (Castellanos-Galindo et al., 2021). This difficulty has prevented so far corroborating most of the information from modeling studies that have highlighted the unique characteristics of the mangrove areas in this coast (e.g., Hutchison et al., 2014; Rovai et al., 2018).

To validate previous remote sensed measurements (i.e., Simard et al., 2019) and to assess the complementarity of different techniques, we compare here mangrove tree height and AGB measurements from ground-based inventories, drone imagery and radar data (from the Shuttle Radar Topography mission-SRTM and the TerraSAR-X add-on for Digital Elevation Measurement-TanDEM-X) taken from a remote area in the Colombian Pacific coast recognized to contain the tallest mangroves in the American continent. The combination of field-based methods (tree DBH and height measurements and drone photogrammetry) with existing SRS products in the most extreme range of mangrove canopy height (∼50 m) allows for the first time: (1) the validation of previous models that highlighted the ecological value of mangroves in this region and (2) the recognition that these areas are in need of urgent protection due on-going localized alterations.

MATERIALS AND METHODS

Study Site

The Colombian Pacific coast encompasses ca. 1,500 km in the tropical eastern Pacific biogeographic region that extends from the Gulf of California, Mexico to northern Peru. Almost 2/3 of this coastal region is dominated by mangroves, with ∼80% of the total mangrove area of Colombia (Mejía-Rentería et al., 2018). The whole coast presents annual precipitations of > 2,000 mm, reaching in some areas 8,000 mm yr^{-1}. This extreme precipitation translates into year-long low water salinities (<30) and permanent brackish areas inside bays, small deltas and tidal channels in what is identified as the Eastern Pacific fresh pool (Alory et al., 2012). The tidal regime is semi-diurnal, meso- to macro-tidal with amplitudes during spring tides greater than 4 m (Correa and Morton, 2010). These tidal conditions indicate that mangroves and mudflats are completely exposed at low tides and completely inundated at high tide (during both neap and spring tides).

Nariño, the southernmost coastal Department along the Pacific coast of Colombia, borders the neighboring Esmeraldas province of Ecuador to the South, and the Colombian Cauca Department in Colombia to the North. Nariño alone contains 46% of the total mangrove forests of the country (Mejía-Rentería et al., 2018). This tectonically active coast is characterized by the presence of two high water discharge and dynamic deltas (Mira and Patía; Restrepo and Cantera, 2013) and an almost uninterrupted mangrove belt. The coast is also under-developed in terms of infrastructure with the presence of small villages (<5,000 inhabitants) scattered in a mangrove-dominated landscape. Nariño contains the largest mangrove marine protected area (MPA) of the whole tropical eastern Pacific (the Sanquianga National Park with ∼80,000 ha) and a more recently created mangrove MPA bordering Ecuador (Distrito Nacional de Manejo Integrado Cabo Manglares Bajo Mira y Frontera). North of Sanquianga MPA lies the mouth of the Iscuandé River, an estuarine area dominated by extensive mangrove forests and small human settlements that constitute the Afro-Colombian community council of Esfuerzo Pescador. The water discharge of the Iscuandé River is 6.71 km^3 yr^{-1} (Restrepo and Kjerfve, 2004).

Field Sampling

In November 2019 (6th–8th), three different mangrove sites in the northern coast of the Nariño Department of the Colombian Pacific coast were visited (**Figure 1**). The sites are located near the mouth of the Iscuandé River, an estuarine area completely dominated by mangrove forests, tidal creeks and mud-sand flats that emerge during low tides. Circular plots of 15 m-diameter were established at each site. Ground measurements within these plots included the identification of tree species, diameter at breast height (DBH) and top height of selected trees. While DBH measurements were generally made around the trunk of each tree at 1.3 m above the ground surface, several measurements were made above the high prop roots of *Rhizophora* spp. (e.g., **Figure 2**). To measure DBH, a diameter tape was wrapped around the girth of each tree and a diameter measurement read was made to the nearest 0.1 cm. Heights of selected trees within each plot were collected with the aid of a Nikon Forestry Pro II laser hypsometer. There trees were then geolocated using a handheld GPS (Bad Elf GNSS Surveyor).

Allometric Equations

For each of the plots surveyed, we used the DBH values and two allometric equations to determine tree biomass per hectare. First, we used the allometric equation for *Rhizophora mangle* developed by Fromard et al. (1998) to estimate tree biomass that only takes into account DBH:

$$AGB = 0.1282 \times D^{(2.6)}$$

We also used the pantropical tree allometric model to estimate AGB proposed by Chave et al. (2014) which takes into account tree height, DBH and wood specific gravity:

$$AGB_{est} = 0.0673 \times \left(\rho D^2 H\right)^{0.976}$$

where D is DBH in cm, H is tree height in m and ρ is wood specific gravity in g cm^{-3}. ρ was computed as the mean value of the wood specific gravity values available for *Rhizophora* spp. in the Central and South America from the Global wood density database (Chave et al., 2009; Zanne et al., 2009).

To implement the pantropical tree allometric model of Chave et al. (2014), we back-calculated the heights of the trees for which we only had DBH. For that we fitted, a Log-Log model of DBH:H with 26 trees for which we had both DBH and Heights (see **Supplementary Figure 1**). Total living carbon was then calculated using the relationship total living biomass:living carbon of 1:0.464 (Kauffman et al., 2011, 2020).

Drone Surveys

Drone surveys in the direct vicinity of each plot were carried out at low tide with calm winds. The drone patches surveyed were ∼3 ha in size at each site and always covered the plot areas where ground measurements were taken. Low-altitude (100 m) aerial images were acquired using two UAS, DJI Mavic Pro and Mavic 2 Pro. These drones are equipped with integrated photo cameras. DJI Mavic Pro has a FC220 camera, with a sensor 1/2.3" CMOS (effective pixel:12,35 Megapixel), focal length of 5 mm, a pixel size of 1.7 × 1.7 μm and a resolution of 4,000 × 2,250 pixels. The Mavic 2 Pro has a L1D-20c camera with a sensor 1" CMOS camera (effective pixel:20 Megapixel), focal length of 10 mm, a pixel size of 2.41 × 2.41 μm and a resolution of 5,472 × 3,648 pixels. Photos are geotagged by the DJI integrated GPS/GLONASS location system with a horizontal and vertical precision of up to ± 1.5 and ± 0.5 m, respectively.[1] Flights were performed in automated mode and were programmed using the commercial web app "Drone Deploy." Due to the nature of the environment, characterized by dense vegetation with lack of large gaps exposing the terrain within the forest, it was not possible to easily place and collect ground control points (GCPs). Although few small gaps are present, they are not easily accessible and lack

[1] www.dji.com/mavic

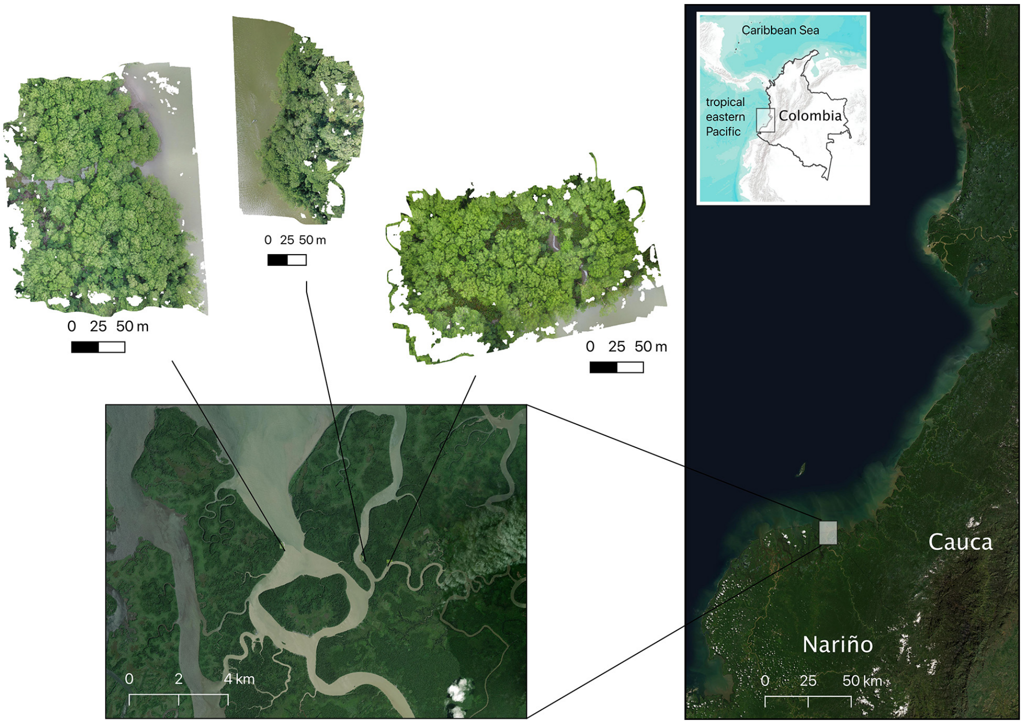

FIGURE 1 | Location of the mangrove sites visited in the southern Colombian Pacific coast (Iscuandé River mouth, Nariño Department; Imagery ©2021 Google, Maxar Technologies, Data SIO, NOAA, U.S. Navy, NGA, GEBCO, Landsat/Copernicus, Imagery ©2021 Terrametrics, Map data ©2021). Three mangrove patches were surveyed in the estuarine complex formed close to the mouth of the Iscuandé River: from left to right, sites are known as: "La Punta del Codo," "La Rotura," and "Madrid."

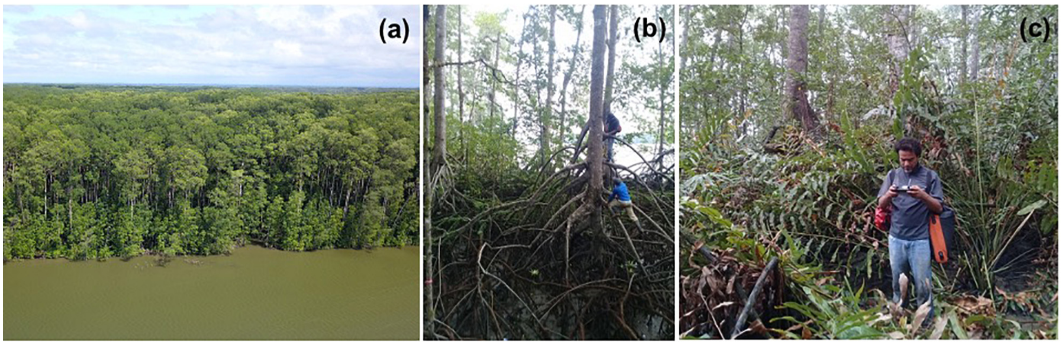

FIGURE 2 | Field surveys performed in mangroves of the southern Colombian Pacific coast (Iscuandé, Nariño). **(a)** Aerial view of one of the sites surveyed (La Punta del Codo); **(b)** measuring DBH of a gigantic *Rhizophora* spp. tree **(c)** piloting a DJI Mavic consumer-grade drone inside a mangrove forests surrounded by the golden leather fern *Acrostrichum aureum*.

the necessary sky view to operate GNSS and collect GCPs with acceptable accuracy.

The acquired images were analyzed and processed using the software Agisoft Metashape.[2] Metashape is based on Structure from Motion (SfM) (Ullman, 1979) and Multi-View Stereo reconstruction (MVS) methods (Scharstein and Szeliski, 2002; Seitz et al., 2006). For a comprehensive description of the SfM method implemented in Metashape, the reader is referred to Westoby et al. (2012). Metashape gives as outputs an orthorectified photomosaic (orthomosaic) and a Digital Surface Model (DSM) from nadir photos collected during the flight. The output models were georeferenced in Metashape to the WGS84 datum using camera positions. The results of the photogrammetric suite were used in the three locations to estimate different environmental variables. Orthomosaics with 2 cm/pixel resolution allowed detecting the heterogeneity of vegetation and the presence of opportunistic species associated to mangrove degradation. DSMs were

[2]www.agisoft.com

used to estimate the heights of trees within the *in situ* sampling areas. Since the ground was not visible within the forest, we used the elevation of mud-sand flats that emerge during low tides as a reference. The observed DSM was vertically shifted to match the elevation of the mud-sand flats. Consequently, the height of the trees was extracted from the vertically shifted DSM.

Satellite Data

To compare our *in situ* and drone generated data, we used two global RS products generated by Simard et al. (2019). One of maximum mangrove canopy height for the year 2000 with a resolution of ∼30 m that used the Shuttle Radar Topography Mission (SRTM) global digital elevation model (DEM), and the Geoscience Laser Altimeter System (GLAS) global Lidar altimetry products. The second RS product was an aboveground mangrove biomass map that was generated by Simard et al. (2019) linking field-measured biomass–height allometry with SRTM estimates of basal area weighted height (all RS products available at https://daac.ornl.gov/cgi-bin/dsviewer.pl?ds_id=1665).

We additionally compared our mangrove height field-generated data against the high-resolution data product (12 × 12 m) derived from the TerraSAR-X add-on for Digital Elevation Measurement-TanDEM-X mission (Simard unpublished *data*). We performed a similar procedure as with the drone DSMs and adjusted the TanDEM-X DEM shifting it vertically to match the elevation of the adjacent water. For details about use of the TanDEM-X in mangrove canopy height estimations see Lee and Fatoyinbo (2015) and Lagomasino et al. (2016).

The tree height comparisons between the three methods used was done using the ground height measurements taken at the three sites surveyed (n = 21 trees). With the georeferenced location of each of these trees, a vector (∼5 m radius circle) was created in QGIS and using the zonal statistics, height information (mean, maximum and minimum) from the drone, the SRTM and the TanDEM-X rasters were extracted. We fitted linear regressions between *in situ* maximum tree heights and calculated drone, SRTM and TanDEM-X tree heights. In addition, Pearson, determination coefficients and RMSE (root mean squared error) were computed to assess the fit of the regressions.

RESULTS

All surveyed sites were dominated by the red mangrove *Rhizophora* spp. In one of the sites (Madrid; **Figure 3**), the golden leather fern *Acrostrichum aureum*, a mangrove associate, dominated the understory which in many cases reached up to 4 m (see **Figures 2**, **4**). While still difficult to distinguish in the drone orthomosaic, we estimated that at least 5% of the orthomosaic obtained for the Madrid site was completely covereded by *A. aurem (*i.e., gaps with no *Rhizophora* spp. canopies). Moreover, our ground surveys at this site showed that the understory in the remaining 95% of the orthomosaic was completely dominated by this species, which was not captured by the drone orthomosaic (**Figure 4**).

Mean DBH values were very high at the Madrid site in comparison with the other two surveyed sites. The Madrid site was characterized by the largest trees with a mean DBH of 52.4 cm reaching a DBH and height of 96.7 cm and 57 m, respectively. In contrast, the mean DBH at La Rotura and La Punta del Codo sites, were considerably lower (around 20 cm) with maximum tree heights of 50.8 and 53.6 m, respectively. The low mean DBH values indicate the presence of comparatively smaller trees in those two latter sites. Nonetheless, the total AGB was, as expected, dominated by the presence of the largest trees in all sites. Depending on the allometric equation used (Fromard et al., 1998; Chave et al., 2014), the mean AGB across all sites was 862.2 or 626.4 Mg ha^{-1}, respectively. Similarly, mean above-ground mangrove carbon stocks were estimated in 400.1 or 290.7 Mg C ha^{-1} (**Table 1**). These values, especially in the Madrid site do not account for the contribution of *A. aurem* to the total biomass and carbon. The mangrove AGB values derived from the ground data were two to three times higher than those estimated with the SRTM data in Simard et al. (2019). AGB values for the examined plots derived from SRTM, and based on continental scale allometry, ranged from 296.2 to 413.3 to Mg ha^{-1}.

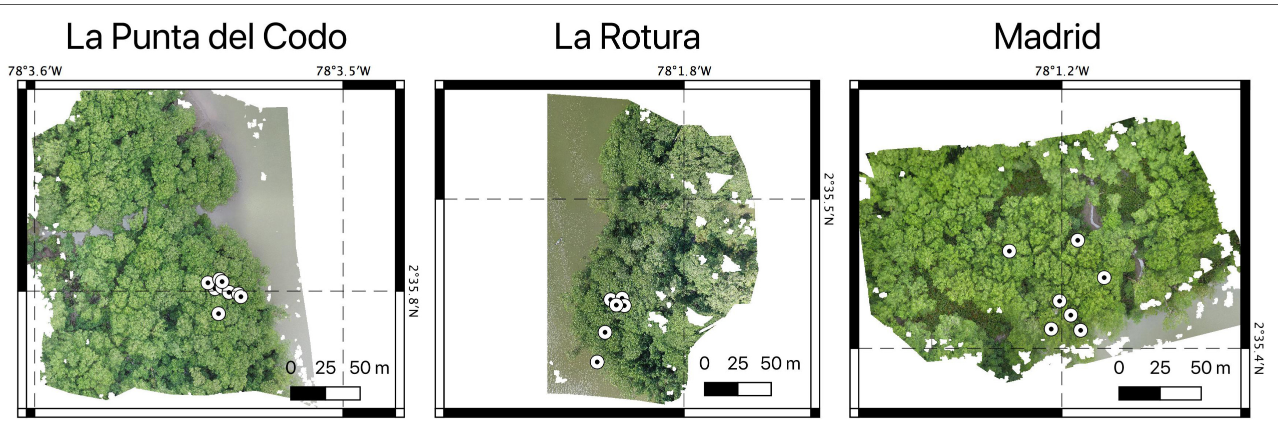

FIGURE 3 | The three RGB orthomosaics obtained with the drone data showing the geo-referenced location of the trees measured for height using the Nikon Forestry Pro II laser hypsometer.

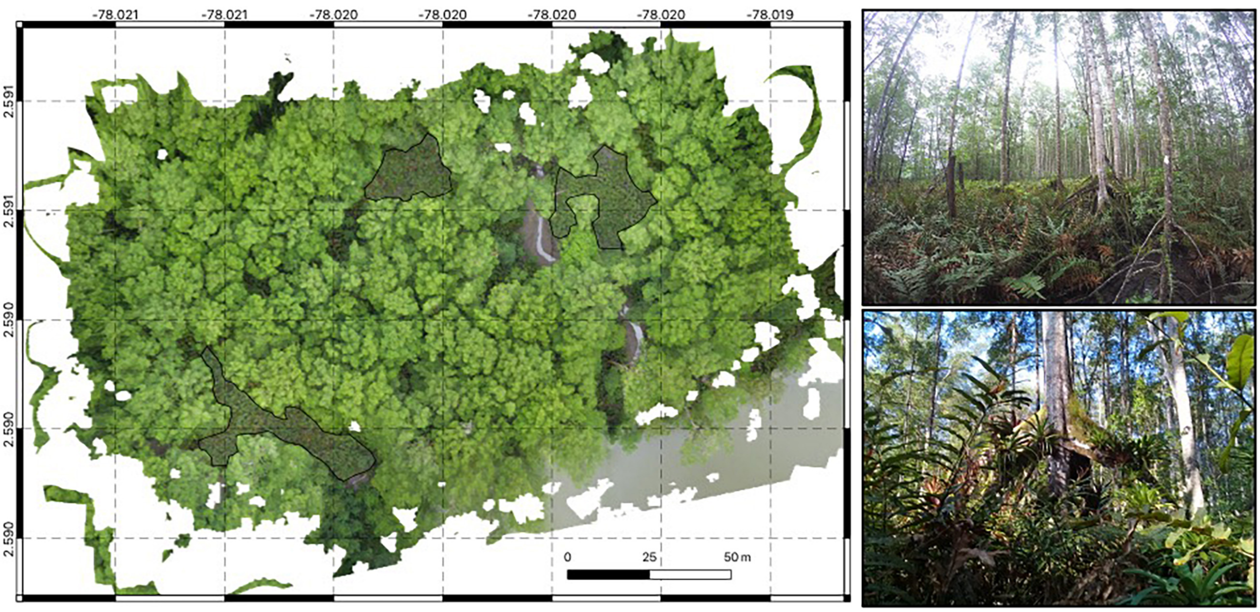

FIGURE 4 | Drone orthomosaic of one of the mangrove sites surveyed in Iscuandé, Colombian Pacific. The transparent delineated areas in the orthomosaic show regions entirely covered by the golden leather fern *Acrostrichum aureum*. Pictures on the right show how the understory of these areas is dominated by *A. aureum*.

TABLE 1 | Summary statistics for the field measurements (diameter at breast height-DBH, maximum height, basal area, and tree density) taken at three mangrove sites in Iscuandé, Colombian Pacific coast.

Site	Samples (n)	Mean DBH (cm)	Max DBH (cm)	Min DBH (cm)	Mean tree height (m)	Max tree height (m)	Basal area (m^2 ha^{-1})	Tree density (trees ha^{-1})	Chave et al. (2014) Pantropical equation		Fromard et al. (1998) Equation	
									AGB (Mg ha^{-1})	C storage (Mg ha^{-1})	AGB (Mg ha^{-1})	C storage (Mg ha^{-1})
La Punta del Codo	27	22.7	73.9	5.0	47.0	53.6	28.4	382.0	848.0	393.5	579.4	268.8
La Rotura	33	19.5	92.0	5.0	46.1	50.8	33.4	466.9	902.1	418.6	667.6	309.8
Madrid	22	52.4	96.7	14.1	47.2	56.7	30.1	114.3	836.4	388.1	632.3	293.4

DISCUSSION

We conducted a field campaign specifically to validate and study the regions with the tallest mangrove forests in the Americas (Simard et al., 2019). Field measurements of tree DBH and height were used to validate drone- and SRTM- derived maps of mangrove canopy height, confirming the findings of Simard et al. (2019). The tallest field-measured tree was 57 m, the drone-derived canopy height revealed maximum heights of 57.6 m and the SRTM data for these sites showed maximum canopy heights of 54.3 m. While our analysis is performed in forest stands that may not be representative of the entire region, we show that *in situ*, drone and spaceborne data provide accurate results in the most extreme range of mangrove canopy height. All the three data sources corroborate the existence of exceptionally tall mangrove forests in this region of the Western American tropics. However, each of the methods used to estimate tree height data presented challenges and associated errors (**Figure 5** and **Table 2**). As revealed in previous modeling studies (e.g., Rovai et al., 2016; Hamilton and Friess, 2018; Rovai et al., 2018), *in situ* measurements confirmed that these sites are on the upper range of mangrove biomass and carbon per area in the world.

Field tree height data was collected with a laser range finder (Nikon Forestry Pro II laser hypsometer). In a recent study, Saliu et al. (2021) indicated that this method presents the lowest amount of error (8%) among a variety of methods used to measure mangrove tree heights in Malaysia. Larjavaara and Muller-Landau (2013) have also identified that both methods (tangent and sine) available when using the laser range finder can incur in high random errors and underestimations in moist tropical forests. A few aspects complicate mangrove tree height calculations with this instrument. For example, in highly dense forests, the identification of the ground and the top of the tree crown can be challenging and thus is not exempt from errors. For a mangrove system in the Colombian Caribbean, Simard et al. (2008) calculated that the random error of tree height measurements in the field can be ∼10%. In the forests examined

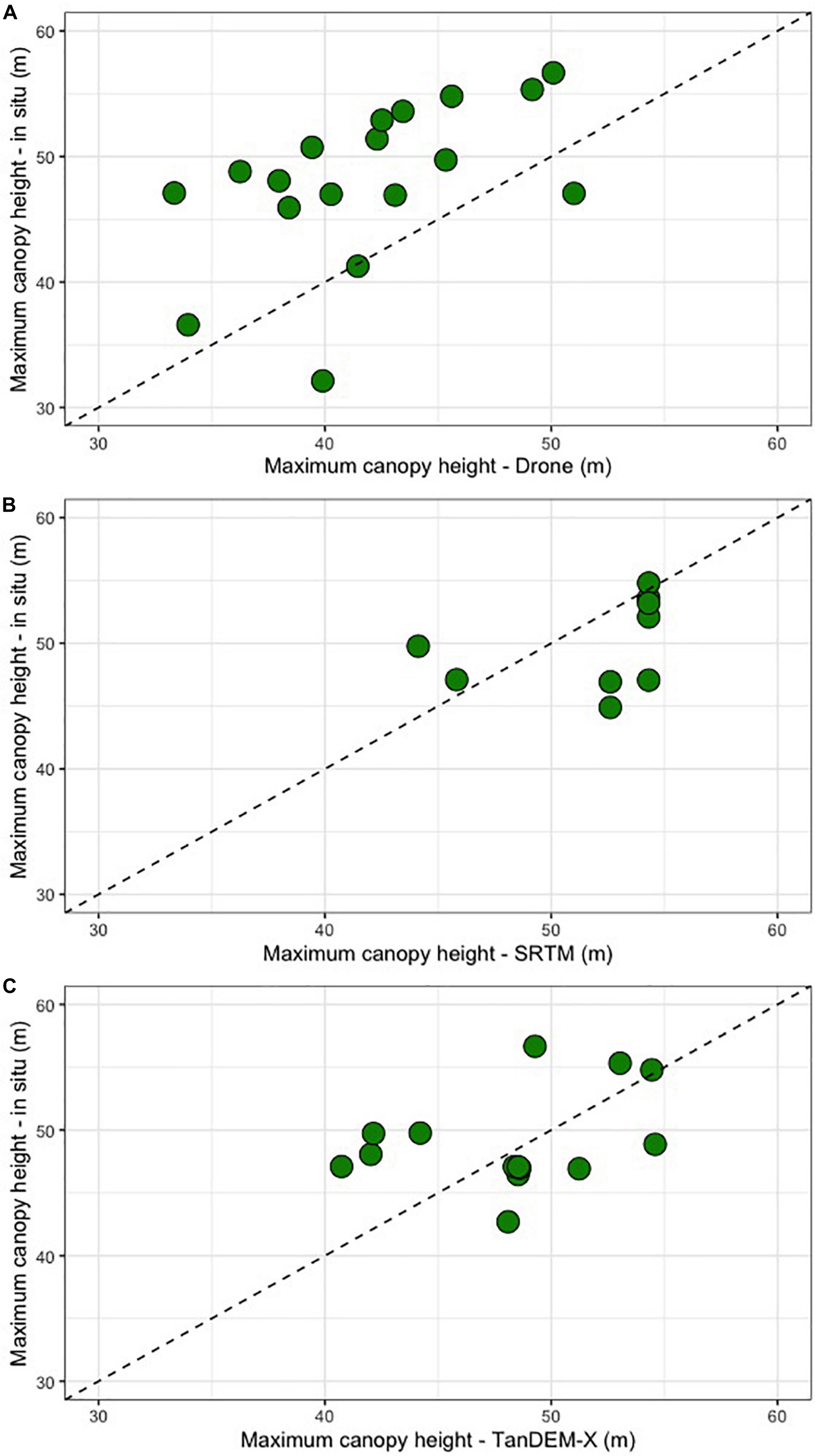

FIGURE 5 | Correlation between different methods to calculate mangrove tree heights in three different sites in Iscuandé, Colombian Pacific. **(A)** Scatter plot of *in situ* Height$_{max}$ vs. Drone Height$_{max}$; **(B)** scatter plot of *in situ* Height$_{max}$ vs. SRTM Height$_{max}$; **(C)** scatter plot of *in situ* Height$_{max}$ vs. TanDEM-X Height$_{max}$.

here, the presence of a dense understory in some sites plus the stilt root nature of *Rhizophora* trees also complicate the identification of the ground with the laser range finder. While we consider these *in situ* measurements to be the most reliable estimates of tree height, we acknowledge the potentially significant uncertainty associated with this method.

TABLE 2 | Summary statistics showing results of linear regressions and Pearson correlations for the comparison among the three different tree height measurements.

Methods compared	Pearson coefficient (*r*)	Determination coefficient (R^2)	RMSE (m)
In situ vs. drone	0.25	0.06	5.69
In situ vs. SRTM	0.34	0.12	3.18
In situ vs. TanDEM-X	0.34	0.12	3.55

Mangrove Above-Ground Biomass of the Iscuandé Region in the Global Context

The mean mangrove AGB values obtained here using the *in-situ* data are extremely high varying between 862.2 and 626.4 Mg ha^{-1} (depending on which allometric equation is used). These values are located in the upper range of the recent global synthesis by Rovai et al. (2021) and also correspond to the upper range values recently estimated by Trettin et al. (2021) for a mangrove forest in Gabon where mangrove trees of up to 60 m can be found. Our estimated mean AGB values are generally higher than those obtained using the AGB values calculated by Simard et al. (2019) for the same areas. This result warrants a re-examination of the relationship between SRTM data and *in situ* inventories considering a larger dataset than the one currently considered in our study. Likewise, it is important to consider that our plots represent very tall mangrove forests within the Iscuandé area. A more representative AGB (and carbon) estimate would need a sampling design that considers the whole range of mangrove type mosaics and patch heterogeneity that are present in this large area (e.g., like the recent work of Trettin et al., 2021 in Gabon). Such a study could yield slightly lower mean AGB values for these mangrove forests but would also contribute to more accurate numbers on carbon stocks of these areas.

Advantages of Drone-Derived Data to Detect Mangrove Degradation

The drone derived data (orthomosaics and DSMs) of these three mangrove sites provided sufficiently detailed information (<4 cm pixel^{-1} resolution) to detect landscape features that are not captured with the ground surveys nor with the SRTM or the TanDEM-X products. Especially important are (1) the heterogeneity in canopy heights observed and also (2) the detection of canopy gaps that are the result of possible deforestation (i.e., selective logging). On one hand, the drone-derived DSMs (3.5 cm pixel^{-1} resolution) recorded highly variable tree heights within each of the 3 ha patches. On the

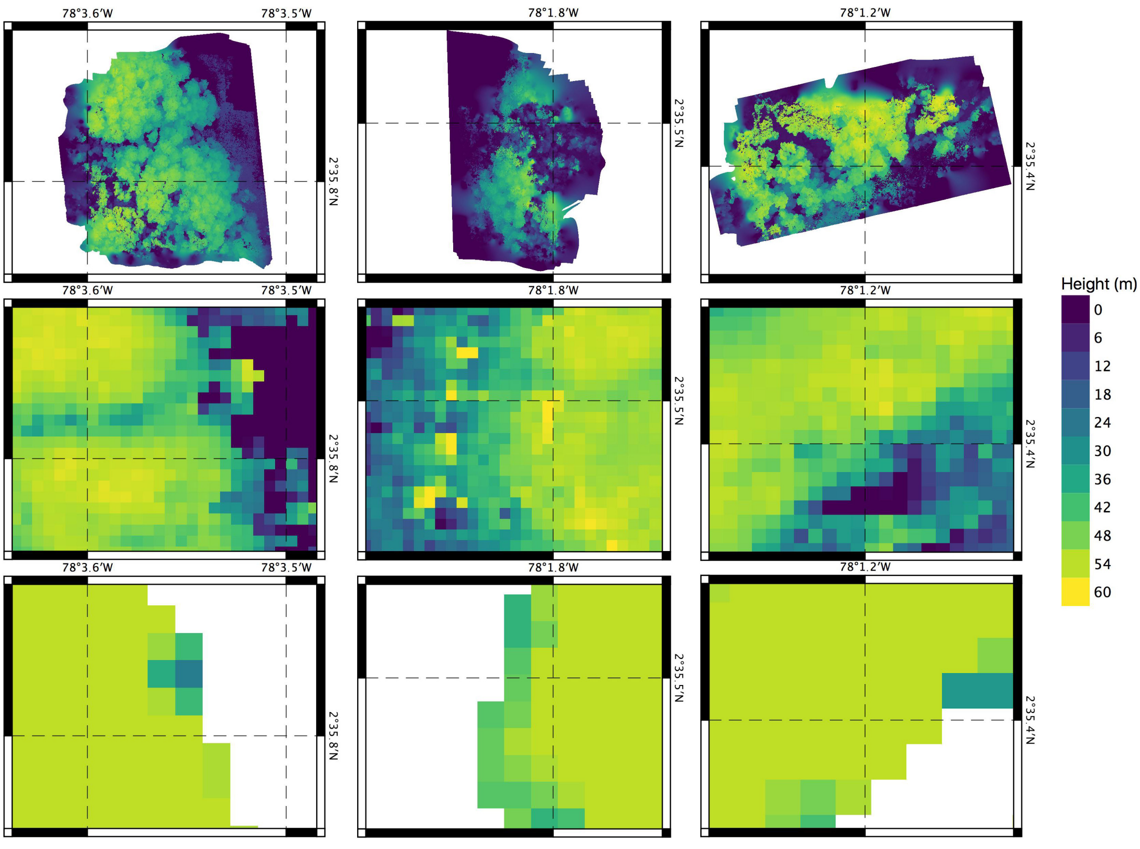

FIGURE 6 | Comparison of altimetry products derived from surveys in mangroves of Iscuandé (Colombian Pacific coast). Mangrove patches correspond to "La Punta del Codo," "La Rotura" and "Madrid" from left to right. The upper row shows the vertically shifted DSMs obtained from the drone data. The middle row depicts canopy heights from the TanDEM-X product (Simard unpublished; © DLR 2020) The lower row corresponds to canopy heights from the Simard et al. (2019) dataset.

other hand, the pixels SRTM and TanDEM-X products may include tree crowns of short and tall trees or ground areas, thus these elevation of these mixed pixels represent overall canopy height rather than the height of the tallest tree. This effect may be particularly important in this study (**Figure 6**), with tree heights measured near forest edges along the shore or around canopy gaps.

On the other hand, the drone orthomosaics were sufficiently detailed to detect the presence of gaps within one of the mangrove patches that were dominated by the golden leather fern *A. aureum*, an opportunistic species associated to mangrove degradation (Madrid patch in **Figure 4**). *Acrostichum aureum* is a highly resistant fern able to colonize the understory of mangrove areas when facilitated either by the presence of natural canopy gaps (see Amir, 2012; Amir and Duke, 2019) or by man-made gaps produced after forest logging. The fern is commonly associated to deforestation and is able to rapidly colonize mangrove areas that have been disturbed, a process that has been defined as cryptic (mangrove) ecological degradation (Dahdouh-Guebas et al., 2005; Blanco et al., 2012). Once established, *A. aureum* modifies the geomorphology and hydrology, impeding natural regeneration and hindering animal movement (Medina et al., 1990; Biswas et al., 2018). Other aspects of the effects of this fern on ecosystem processes within the mangrove forests are poorly understood. The information provided by the drone products on *A. aureum* can be valuable as it provides evidence at the scale of 10 s of hectares on loss of mangrove ecosystem health that would otherwise be unnoticed when using coarser SRS products. These highly detailed drone products can also aid in the development of complexity indexes, such as those proposed in Blanco et al. (2001), that could be used to characterize health status and degradation in mangrove forests. Being able to identify the understory vegetation (e.g., *A. aureum* in this case) of a mangrove area with the drone products implies also that the contribution of this vegetation to the ecosystem biomass and carbon storage, which could become very high, could be accounted for. However, it is unclear how persistent in time *A. aureum* can become in this region. Surprisingly, allometric equations and information on the carbon content and establishment dynamics of this opportunistic fern are scarce and needed.

The mangrove areas examined here, still contain well developed forests. However, historical mangrove timber exploitation has occurred and continues to occur. Although we do not have data on the magnitude of recent and on-going logging, the local communities identify this activity as an increasingly pressing factor on the surrounding mangrove ecosystems. Moreover, unaccounted as a mangrove degradation source remains the effects that the anti-narcotics national program for eradication of coca crops that the Colombian government undertook in the 2000s with aerial aspersion of glyphosate (and currently seeks to continue). One of the few accounts on the use of herbicides in mangrove forests is that of the severe effects that its use caused on Vietnam forests during the second Indochina war (Westing, 1983). Anecdotical information from locals in Iscuandé reported on the effects that glyphosate had on native flora and fauna around mangroves in the past. The localized effects that both timber logging and glyphosate could have on these exceptionally well-developed mangroves could be studied with the finer resolution products from drones.

Implications for Mangrove Carbon Monitoring in Remote Areas

Our study reveals significant challenges in collecting field and drone data in these mangrove areas. These can be categorized as logistical and geometrical. The access to these sites requires careful travel planning that includes site accessibility by plane and boat in a highly dynamic setting driven by the large tidal range, and close collaboration with local authorities and organizations. The latter facilitates, not only safe passage to sites, but gathering of a wealth of knowledge about local landscapes, cultural practices and customs related to mangroves and water management (see Oslender, 2016). On the other hand, the geometrical challenges are related to the complex terrain with mangroves roots, litter, and mud that constrains the possibility to collect tree heights in a systematic and time-efficient way. To accurately process the drone data, collecting ground control points-GCPs (ideally with a Real-time kinematic-RTK GPS) is desirable. However, identifying sufficiently clear gaps to place those GCPs inside mangrove forests can prove an unachievable task. Ways to overcome this limitation could include using drones that have incorporated an RTK GPS (although this technique does not completely exclude the need of GCPs) and/or using adjacent ecosystems within the mangrove forest (e.g., mudflats) where placing GCPs is feasible.

Zeng et al. (2021) suggests that mangrove blue carbon projects in Colombia may only be profitable in the southern Colombian Pacific coast, the area where this study was conducted. Our results, therefore provide valuable assessment methodologies and an initial *in situ* assessment of carbon stocks for potential blue carbon projects in an area that has been highlighted by modeling studies as a blue carbon hotspot in the world with much need of effective conservation actions. The techniques are complementary and may represent the three tiers for carbon reporting, whereby the drone-derived canopy height maps, calibrated with local *in situ* measurements, provides affordable and reliable Tier 3 estimates of carbon stocks at the project level. These detailed estimates can in turn help calibrating the coarser Tier 1 and Tier 2 assessments. Additionally, the combination of drone and ground survey data can provide a way to detect and monitor small-scale ecosystem degradation symptoms that so far are not possible to detect with the SRS products currently used for mangrove mapping. This can bring a more detailed understanding of landscape processes occurring within these vulnerable ecosystems that so far have been excessively focused on coarse area measurements.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author/s.

AUTHOR CONTRIBUTIONS

All authors contributed to the article and approved the submitted version.

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Natural and Anthropogenic Variation of Stand Structure and Aboveground Biomass in Niger Delta Mangrove Forests

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Mangrove forests are important coastal wetlands because of the ecosystem services they provide especially their carbon potential. Mangrove forests productivity in the Niger Delta are poorly quantified and at risk of loss from oil pollution, deforestation, and invasive species. Here, we report the most extensive stem girth survey yet of mangrove plots for stand and canopy structure in the Niger Delta, across tidal and disturbance gradients. We established twenty-five geo-referenced 0.25-ha plots across two estuarine basins. We estimated aboveground biomass (AGB) from established allometric equations based on stem surveys. Leaf area index (LAI) was recorded using hemispherical photos. We estimated a mean AGB of 83.7 Mg ha^{-1} with an order of magnitude range, from 11 to 241 Mg ha^{-1}. We found significantly higher plot biomass in close proximity to a protected site and tidal channels, and the lowest in the sites where urbanization and wood exploitation was actively taking place. The mean LAI was 1.45 and ranged fivefold from 0.46 to 2.41 and there was a significant positive correlation between AGB and LAI (R^2 = 0.31). We divided the plots into two disturbance regimes and three nipa palm (*Nypa fruticans*) invasion levels. Lower stem diameter (5–15 cm) accounted for 70% of the total biomass in disturbed plots, while undisturbed regimes had a more even (∼25%) contribution of different diameter at breast height (DBH) size classes to AGB. Nipa palm invasion also showed a significant link to larger variations in LAI and the proportion of basal area removed from plots. We conclude that mangrove forest degradation and exploitation is removing larger stems (>15 cm DBH), preferentially from these mangroves forests and creates an avenue for nipa palm colonization. This research identifies opportunities to manage the utilization of mangrove resources and reduce any negative impact. Our data can be used with remote sensing to estimate biomass in the Niger Delta and the inclusion of soil, leaf properties and demographic rates can analyze mangrove-nipa competition in the region.

Keywords: mangrove, aboveground biomass, leaf area index, disturbance, stand structure (tree size diversity), Niger Delta

INTRODUCTION

Mangroves are very productive ecosystems due to their tropical coastal location (Alongi, 2009). These ecosystems provide a range of ecosystem services, including provisioning services (e.g., fisheries and fuelwood), regulatory services (e.g., carbon storage, nutrient cycling and shoreline protection), and cultural/aesthetic values (McLeod and Salm, 2006; Bouillon et al., 2008; Feka and Ajonina, 2011; Kauffman et al., 2011; Mukherjee et al., 2014; Friess, 2016). Mangroves act as a valuable carbon sink contributing ∼15% to coastal sediment storage of carbon, despite inhabiting up about 0.5% of the world coastal area (Alongi, 2014). However, mangrove ecosystems are threatened by deforestation and contribute ∼10% of the total global deforestation emissions of CO_2 (Donato et al., 2011). The relevance of carbon storage in mangrove sediments and deforestation rates has made mangrove an essential focus for climate change mitigation through conservation and reforestation projects, for instance under the Reducing Emissions from Deforestation and forest Degradation (REDD+) program (McElwee et al., 2017).

Nigeria's coastal zone includes lagoons, deltas and estuaries comprising mangrove forests and sandy beaches, with a semi-diurnal tidal regime. The mangrove wetland in Nigeria is ranked the fifth country with the largest mangrove area globally (Giri et al., 2011; Hutchison et al., 2014; Lucas et al., 2014). The Niger Delta contains about 60% of Nigerian mangroves, measuring about 801,774 ha in 2017 (FAO, 2005; Fatoyinbo and Simard, 2011; Nwobi et al., 2020). Mangrove forests have relatively low species diversity in Nigeria, with only three genera which include *Rhizophora* sp. (red mangrove), *Laguncularia racemosa* (white mangrove) and *Avicennia* sp. (black mangrove) (Food and Agriculture Organization, 2007). Pristine mangrove forests in Nigeria serve as important sources of seafood including shellfish, finfish and nursery grounds for these organisms (Feka and Ajonina, 2011). These aquatic organisms are also vital indicators of undisturbed mangrove ecosystems in coastal Nigeria (Amadi et al., 2014). Similarly, the presence, absence or abundance of specific floral species are indicators of mangrove perturbation (Mmom and Arokoyu, 2010). However, increasing population, resultant development and industrial activities are changing this valuable ecosystem. Coastal development, aquaculture expansion and over-harvesting have led to a 30–50% reduction in global mangroves over the past 50 years (Kauffman et al., 2011). Loss of mangrove wetlands in the Niger Delta is prominently due to oil spills, land reclamation for housing, road, electricity power lines, port development and dredging waterworks (Feka and Ajonina, 2011). Local communities depend on mangrove cutting for fuelwood and commercially for sale in the Niger Delta. Mangroves also provide wood products to the wood industry in Nigeria; however, this practice is unsustainable and threatens mangrove forests sustainability (Kinako, 1977). Unchecked logging of mangrove trees leads to a reduction in mangrove extension and has been linked to the expansion of the invasive *Nypa fruticans* within the Niger Delta (Okugbo et al., 2012; Global Invasive Species Database (GISD), 2015). Information on patterns of mangrove loss is sparse, but vital to support conservation measures. Limited research in mangrove forests in Nigeria is primarily due to social unrest, restricted access and security. These hindrances to mangrove forest research have resulted in limited data on mangrove forest structure in the Niger Delta. Mangrove research in Nigeria is restricted to community structure relationship with soil properties, carbon dynamics and productivity, remote sensing of forest area and remediation/ remediation (Ukpong, 1994, 2000b; James et al., 2007, 2013; Jackson, 2011; Fatoyinbo and Simard, 2013; Edu E.A. et al., 2014; Edu E.A.B. et al., 2014). One viable option for mangrove research in Nigeria include the use of remote sensing due to the challenge of field surveys in the region (Nwobi et al., 2020). However, remote sensing analysis relies on sound ground data for calibration, and hence fieldwork is still an important requirement.

Field estimates of stand structure [e.g., basal area (BA) and stem size], canopy properties [e.g., leaf area index (LAI)] and aboveground biomass (AGB) are needed to build a baseline to monitor mangrove forest change, develop restoration plans, and support calibration of remote sensing data. For instance, BA is the basis for monitoring the removal of mangrove stands from logging activities (Ngoc Le et al., 2016). Both LAI (Clough et al., 1997) and change in AGB (Alongi, 2009) can be used as indicators of net primary productivity. In-situ measurement of ABG and LAI over a forested landscape are also vital for integration with satellite imagery forest assessment with both optical and radar data (De Kauwe et al., 2011; McNicol et al., 2017). However, in order to monitor local disturbance such as targeted logging or determine the expansion rate of invasive species, a fine scale of observations need to be adopted in mangrove monitoring programs.

Selective harvesting in natural forests is a subtle activity being carried out by local communities in the Niger Delta. This wood exploitation can result in the change in stand size and canopy structure (Walters, 2005b), since this extraction targets particular tree size classes when harvesting wood. These tree sizes exploited are called the target tree size, which is the most economical range of tree size harvested in order to maximize profit. For mangrove forests located along creeks, target stems are those with maximum harvestable tree sizes that allow efficient water transport to the point of sale. The target size class depend on the type of forest, wood species,the distance of forest stands from the point of sale, type of harvesting tool, the harvester gender and transportation method (Allen et al., 2001; Walters, 2005a; Feka and Ajonina, 2011). On Kosrae, Micronesia they reported a favorable species for harvesting as *Lumnitzera littorea* and a target diameter size of 10–30 cm (Allen et al., 2001). The change in mangrove stand size structure has a direct effect on stand biomass by altering the relative contribution of different stem sizes to the total forest AGB (Feka and Ajonina, 2011). Selective harvesting can also result in light gaps which are prominent natural process in mangrove forests. Forest gaps can be created from natural (hurricane or lightning) or anthropogenic (wood exploitation) disturbance (Amir and Duke, 2009). Selective harvesting and the resulting light gaps are detrimental to mangrove forests of the Niger Delta because of the presence of the alien invasive nipa palm

(*Nypa fruticans*). Light gaps from tree cutting creates an avenue for either colonization by invasive species or reestablishment of mangrove species (Schnitzer et al., 1991; Harun Rashid et al., 2009; Potin, 2013). Mangrove and nipa palm shrubs growing together within light gaps formed by logging are common features in the Niger Delta.

This is the first report on mangrove biomass in the Niger Delta that spans two states in Nigeria and covers a wider area from the Niger Delta creeks in the Rivers State to the Imo river estuary (IRE) in the Akwa Ibom State. Previous biomass surveys were restricted to one location using small plot size (Nwigbo et al., 2013; Numbere and Camilo, 2018). This is also the first report on mangrove biomass spatial distribution across the Niger Delta from the ocean, tidal channel and settlement areas. Previous reports have studied mangrove distribution in relation to soil patterns but there is no report on AGB in the delta proper (Ukpong, 1994, 2000a,b). We also provide a first step to monitor mangrove productivity by establishing a relationship between the forest canopy structure and woody biomass. The invasion of nipa palm in mangrove forests is a subtle issue in the Delta, slowly replacing the natural mangrove stands because of deforestation. Nipa palm was first introduced in the early 1900s to control beach erosion but due to poor management, has been growing unchecked in coastal Nigeria. Nipa palm has continually increased by sevenfold between 2007 and 2017 (11,774 ha), naturally invading mangrove forests after disturbance (Nwobi et al., 2020). Thus, we report the possible effect of wood exploitation on the colonization of nipa on mangrove forests. Nipa palm has been termed a secondary successor to destroyed mangrove forests in the Niger Delta (Ukpong, 2015). Previous reports have estimated area coverage per state (Isebor et al., 2003) and nipa influence in changing habitat (Ukpong, 2015), but there is no assessment on the relationship between nipa invasion and mangrove forest anthropogenic disturbance. We also report here the first largest stem size survey in the Niger Delta and how local disturbance is altering the contribution of these stands to the biomass stock of Niger Delta mangrove forests. The relationship among biomass, disturbance and stand structure in the Niger Delta can inform restoration projects on target stand size and help develop management plan.

The objective in this research is to provide a large survey of the relationships amongst mangrove stand, canopy structure and biomass structural and spatial patterns in the Niger Delta. We address the following questions: **(a)** what is the natural variation of AGB and LAI across the Niger Delta study site? **(b)** What are the effects of anthropogenic factors on the structure of mangrove forests? **(c)** Is mangrove wood exploitation a precursor to nipa palm invasion in the Niger Delta? We hypothesize that higher biomass plots will be closer to tidal channels and farther from human settlements. We also hypothesize a positive linear relationship between AGB and LAI. We test the hypothesis that disturbance is removing mangrove stands with higher diameter at breast height (DBH) size thus altering their contribution to the total forest AGB. Lastly, we test the effect of wood exploitation as a precursor to nipa palm invasion in the Niger Delta. Understanding the interplay of local anthropogenic disturbance and nipa palm invasion on mangrove structure can assist in the management of unsustainable wood harvesting.

MATERIALS AND METHODS

The Niger Delta

The Niger Delta is the largest coastal delta in Africa (**Figure 1A**) and the ninth largest wetland in the world with an estimated area of 19,135 km^2 (Dupont et al., 2000). The delta contains all three-mangrove genera characteristic of the Atlantic coasts: *Rhizophora*, *Avicennia*, and *Laguncularia*. These species are *Rhizophora racemosa*, *Rhizophora mangle*, *Rhizophora harrisonii*, *Avicennia germinans*, and *Laguncularia racemosa* species. Temperature ranges from 21 to 33°C. Mean annual precipitation of 2,436 mm (Amechi et al., 2014) ranging between 1,500 to 4,000 mm (NDDC, 2006). There are two seasons in Nigeria with the highest temperatures between February and April (28–33°C); and lowest temperatures (21–23°C) during the peak of the rainy season between June and September (NDDC, 2006).

Study Area

The study was carried out in two states of the Niger Delta—the Rivers and Akwa Ibom states (**Figure 1B**). We established three mangrove study sites within two estuarine basins: Imo River Estuary (IRE) and Sombreiro River Estuary (SRE). The common feature between all sites was their location in the Niger Delta. The site selection was based on mangrove extent, accessibility, security and management regime. Limited access to mangrove forests in the Niger Delta is the result of security issues ranging from kidnapping, oil bunkering and militancy (Aduloju and Okwechime, 2016). However, we used the global forest height map (Simard et al., 2011) to randomly select a wide range of mangrove biomass values within the selected sites. These three locations are Ete creek, Oproama community, and Kono creek (**Figures 1C–F**). These sites were selected to represent the wide range of mangrove community types and ecosystem disturbance regimes experienced by this deltaic region. Field plots were established where data collection was performed. There has been no report of mangrove structure, biomass, and disturbance within this region.

Ete creek is located in Ikot Abasi Local Government Area (LGA), Akwa Ibom State (**Supplementary Figure 1**). This creek runs from Ikot Akan and empties into the Imo river estuary (IRE) at Ikot Abasi. The major economic activity of this region is fishing. However, commercial fishing has resulted in a shift to lumbering (**Supplementary Figure 1A**). This shift in economic activity has caused a high incident of logging and wood exploitation. There are also two oil wells around the creek with the one occurrence an oil spill during fieldwork, which was evident from the presence oil film along the creek (**Supplementary Figure 1B**). Ete creek is fringed on either side with mangrove forests, which expands into rainforests or farmland. Despite having high logging activity, mangroves in Ete creek are as high as 15 m (**Supplementary Figure 1C**). The landward extent of mangrove forests along this creek is

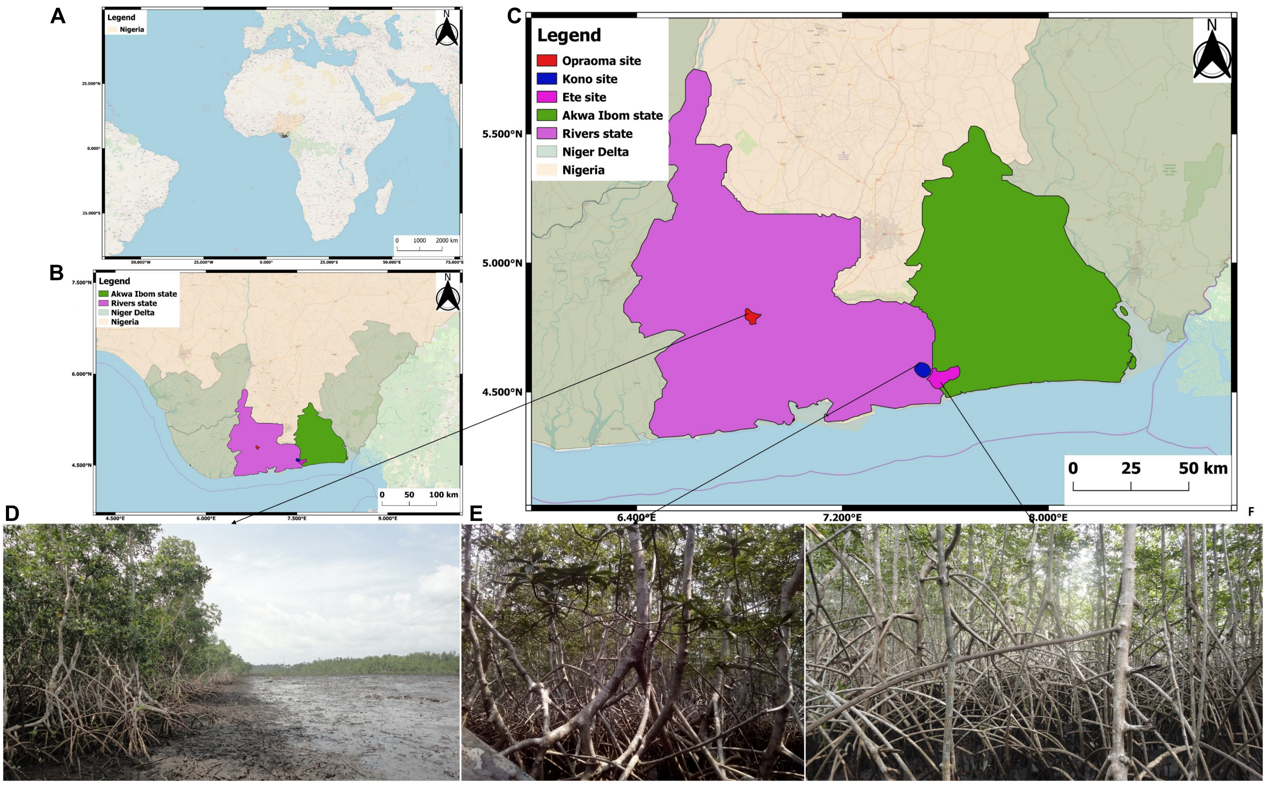

FIGURE 1 | Location of the Nigeria in West Africa **(A)**, the Niger Delta states **(B)**, and field sites location **(C)**. Oproama **(D)**, Kono **(E)**, and Ete **(F)**.

dependent on the economic activity of the locals inhabiting the region including primarily farming with minor activities of fishing and sand mining.

Oproama community is located in Asari Toru LGA, Rivers state (**Supplementary Figure 2**). This community is crisscrossed with creeks that discharge into the Sombreiro river estuary (SRE). The mangrove forests around this community is riverine type that gradually changes into a tropical forest (**Supplementary Figure 2A**). However, urban development is gradually reducing the landward extent of mangrove wetlands which results in complete or partial loss of forest stands (**Supplementary Figure 2B**). This is evident from forest clearance for power lines and road construction (**Supplementary Figure 2C**). The community believe that illegal cutting can result in annoying a deity which is common in the Niger Delta (James et al., 2013). The main economic activity within this region is fisheries and some lumbering in the closest rain forest.

Kono creek is located in the Khana LGA, Rivers state and it empties into the IRE (**Supplementary Figure 3**). There is a protected area along the creek established by the Centre for Environment, Human Rights and Development (CEHRD), where there are dense pristine mangrove forests (**Supplementary Figure 3A**). The communities along the creek protect mangrove forests from any foreign activities that does not involve fishing. However, poor management of the protected area resulted in nipa vegetation expansion. Thus, the creek is fringed with dense nipa vegetation which makes access to mangrove vegetation difficult (**Supplementary Figure 3C**). The main economic activities along this creek are fishing and farming (**Supplementary Figures 3B,E**).

Sampling Strategy

We set up plots to measure stem density, stem size, LAI, and AGB in each study site. Four transects were established in the Oproama community (**O**- field plot designation), Asari Toru Local Government Area (LGA) in Rivers state. Transects included three plots each, but the fourth transect included only two plots due to local community restrictions. This local community value mangrove forests, but there is an ongoing clearance for the development of roads and power lines. One transect with two plots was established in Kono community (**K**), Khana LGA, Rivers state; due to a high level of nipa invasion at the fringes which made mangrove access difficult. We established five transects in Ete (**E**) Kingdom of Ikot Abasi LGA in Akwa Ibom State; two transects with three plots and three transects with two plots each.

Transects established within each site were deployed based on mangrove extension from the tidal channel and variation in mangrove biomass, stand structure and local disturbance. Plots, each 0.25 ha (50 × 50 m), were established within each transect 10 m apart. One of the factors used to define the same size (50 × 50 m) for each plot was because of the use of plot AGB in establishing a relationship between radar sensors and AGB. Varying plot size would compromise the applicability in remote

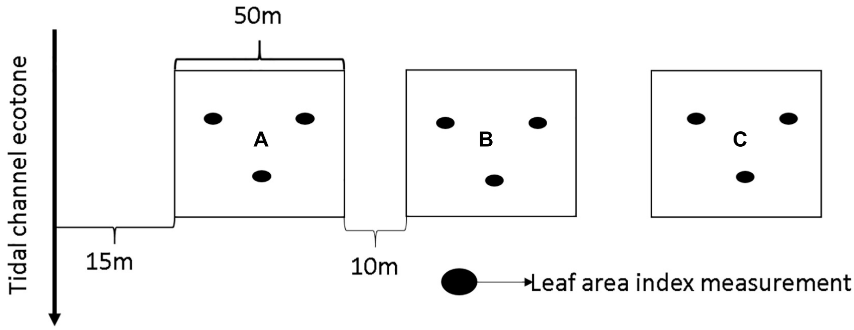

FIGURE 2 | Schematic plot layout for mangrove forest sampling. Letters A–C represent plot distance from tidal channel in all plot identifiers (O, K, E). Letter A represents plot closest to the tidal channel. Plot identifier: O1B (O: location, 1: transect number, B: plot distance from tidal channel).

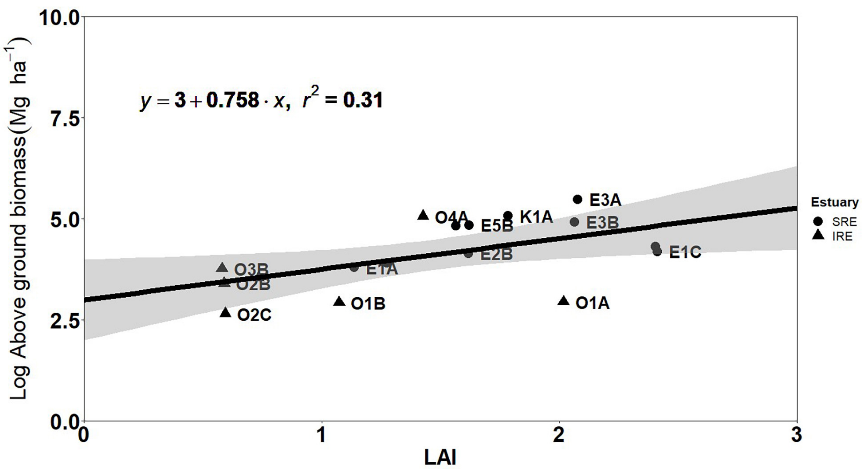

FIGURE 3 | Fitted linear regression of plot of leaf area index (LAI) and aboveground biomass (AGB). Each point represents the plot during the field survey.

sensing analysis. The plot sizes were also chosen to maximize sampling time within very dense mangrove forests. The first plot within each transect was established 15 m from the tidal channel (**Figure 2**). Overall, we established twenty-five sample plots (**SRE-** $n = 11$**, IRE-** $n = 14$**)** from October 2016 to September 2017. During the field survey, we observed mono-generic plots of *Rhizophora* species, with very sparse *Avicennia* species or *Laguncularia racemosa* identified around the plots. We collected global positioning system (GPS) points of the field plots and closest settlement during the field survey using a Garmin eTrex 20x device. The distance of the field plots from distance from the ocean, tidal channel, and closest settlement was also measured. This distance was measured by collecting the GPS locations of the established plots and closest human settlements. The closest settlement was located through communication with the field assistant. We measured the distance between the field plots and settlement from the preferred transportation method (land or water) using Google Earth Pro 7.3.2.5776 (March, 2019). The distance between the field plots and the closest estuarine mouth to the ocean was measured using Google Earth Pro in the direction of tidal flow. We used the ruler tool in Google Earth Pro to trace a path between the settlement and the field plots, either through a road or through the water path. The ruler tool on Google Earth Pro automatically calculated the distance between the settlement and the field plots. We measured the distance to establish the spatial representation of mangrove productivity, population pressure and tidal influence.

We classified the plots into disturbed and undisturbed regimes. We based the classification on visual evidence of disturbance (logging or clearance), mangrove undergrowth and presence of mangrove and nipa undergrowth within forest gaps caused by disturbance. Forest gaps were only considered if there was presence of wood exploitation from logging or clearance. Within each plot, individual nipa palm was counted and the ratio between nipa stand to mangrove stand estimated. Plots were classified into three invasion classes: no invasion (**NI**- 0%), moderate invasion (**MI**- 0–10%) and high invasion (**HI**- >10%) based on the ratio of nipa to mangrove stands within each plot. This division was used to check for the incident of mangrove cutting as a precursor for nipa invasion.

Forest Structure Measurements

We measured, and recorded all trees with a DBH >5 cm within the sampling plot because despite smaller trees can dominate stand composition, we defined a 5 cm minimum as mangrove trees (Food and Agriculture Organization, 1994; Scales and Friess, 2019). DBH was measured at 1.3 m height above the ground, and if the tree branched below 1.3 m, individual stems were measured and counted as one tree (Howard et al., 2014). However, there were unusual cases where we used modified DBH measurement due to the structural complexities of *Rhizophora* spp. (Dahdouh-Guebas and Koedam, 2006; Mahmood et al., 2016). The measured DBH was divided into four size classes: 5–10 cm, 10–15 cm, 15–20 cm and > 20 cm; to account for the DBH range, stand density and stand size structure. Basal area (BA) was calculated by summing over all (n) trees per plot (Equation 1) using DBH in cm (DBH) as described by Cintron and Novelli (1984). BA was reported in meters per hectare (m^2 ha^{-1}).

$$\text{Basal area} = \left(\sum_{i=1}^{n} \pi\left(\frac{DBH_i}{200}\right)^2\right) \div A \tag{1}$$

Aboveground Biomass

We calculated aboveground biomass (AGB, B_{above}) using a common allometric model (Equation 2) for mangrove trees developed by Komiyama et al. (2005):

$$B_{above} = 0.251 \times \rho \times DBH^{2.46} \tag{2}$$

Where ρ = mean wood density of the three indigenous *Rhizophora* species *R. racemosa* = 0.9330 g cm^{-3}, *R. harrisonii* = 0.86 g/cm^3, *R. mangle* = 0.9064 g/cm^3 (Wood Density database Website) and *DBH* is the diameter at breast height in cm (ICRAF, 2010). We used this equation due to the absence of site-specific allometric models. This was the preferred allometric equation over two other allometric equations developed in Cameroun (Ajonina, 2008) and an inventory data from the Niger Delta (Nwigbo et al., 2013). The general allometric equation used was developed from 104 trees of 10 mangrove species with DBH range 5–48.9 cm. The DBH range from this study ranged from 5 to 42 cm falling within the range of the stands used to generate the allometric equation. The inclusion of wood density in the general allometric equation could account for site variation in species. This inclusion is important in estimating the different parts of the tree biomass (Komiyama et al., 2005). The wood density used is similar to values reported by a local study in Nigeria (Adedeji et al., 2013).

We investigated how variations in AGB are related to differences in stand structure, disturbance regime and the distribution of stem sizes. We calculated the proportional contribution of each DBH size class to the total measured AGB within each plot and established the relation between AGB and LAI.

Leaf Area Index

Canopy cover and LAI were from hemispherical photographs taken at three points within each plot (**Figure 2**) using a Nikon D500 camera fitted with a Sigma EX DC HSM (4.5 mm; 1:2.8) circular fisheye lens and Jessop's ultraviolet filter. In order to attain even sky illumination, fish eye photos were captured between 9 am and 3 pm at the peak of exposure and conditions of even skylight and cloud cover (Bequet et al., 2011); when conditions were not too sunny nor too dark. Adequate periods of canopy properties, i.e., dusk and dawn, were not possible as a result of accessibility to field plots. We also waited for minimal wind movement to obtain adequate shutter speed to freeze any foliage movement using small International Organization of Standardization (ISO) settings. We set the aperture at f-9, camera auto exposure so that shutter speed would automatically compensate for changes in ISO. We analyzed hemispherical photographs using the Gap Light Analyzer (GLA) imaging software used to extract forest canopy structure and gap light transmission indices (Frazer et al., 1999). Default threshold levels used in defining leaf and sky features during image analysis were adjusted manually in order to reduce the contribution of sunlight and cloud cover shades to canopy features.

Data Analyses

Linear regression models were used to establish the correlation between biomass (AGB) and canopy characteristics (LAI). Spearman's correlation was used to test for the significant relationship among plot level stand parameters, AGB and canopy properties. We checked the relationship between stem density and AGB, AGB and distance from closest human settlement, distance from coast and AGB, LAI, and AGB. We also performed spearman's correlation to test the relationship between nipa invasion and disturbance. We specifically checked the relationship between nipa stand population and distance from the sea, proportion of basal area removed and LAI variation, nipa stand population and relative contribution of different DBH classes to AGB. We carried out a one-way analysis of variance (ANOVA) to assess significant differences in stand, AGB and canopy properties amongst plots of broadly similar disturbance regimes, DBH size classes and nipa invasion. ANOVA was followed by a Tukey HSD *post-ad hoc* tests. Plot stem density, AGB and LAI were tested independently for differences among sites (Ete, Kono, and Oproama) and disturbance regimes (Undisturbed, Exploited) using a one-way ANOVA. Plot LAI variation, DBH size class 3 contribution to basal area and AGB were tested individually for differences among sites and disturbance regime using a two-way ANOVA. Number of nipa stands within each plot was tested for difference among sites and location using a one-way ANOVA. Plot LAI variation, proportion of basal area removed and DBH size class 3 contribution to basal area and AGB were tested individually for differences among level of nipa invasion (No invasion, Moderate Invasion and High Invasion) using a one-way ANOVA. Where data was not normal, it was log transformed before performing analysis of variance. All data analyses were performed using RStudio version 0.99.491 (R Core Team, 2020).

RESULTS

Forest Inventory

We measured 5,729 mangrove stands (50–560 stand per plot) across 6.25 ha mangrove area. Mean stem density was 903 stems ha^{-1} and mean DBH was 9.9 cm (median = 9 cm). DBH distribution was skewed and unimodal, indicative of a reverse J-shaped DBH distribution (**Supplementary Figure 4**). The highest stem density (4,037 stems ha^{-1}) was recorded in Ete (E3A) and the lowest in O1C (204 stems ha^{-1}). The highest plot mean DBH (13.9 cm) was recorded in Kono (plot K1B) and the lowest (7.5 cm) in Ete (plot E4A). We recorded a mean plot basal area of 8.9 m^2 ha^{-1} and a median of 6.34 m^2 ha^{-1}. The highest basal area (27.24 m^2 ha^{-1}) was recorded in Ete (E3A) and the lowest basal area (1.36 m^2 ha^{-1}) in Oproama (O1C) (**Supplementary Table 1**).

Total stem measurements were divided into stem size classes as follows: Class 1 (5–10 cm) n = 3,709, Class 2 (10–15 cm) n = 1,509, Class 3 (15–20 cm) n = 360 and Class 4 (>20 cm) n = 151. The lowest size classes accounted for a majority (65%) of the stem density (**Table 1**). However, the DBH size class 4 (>20 cm) make up 22% of the total AGB of the region. We also discovered that stem size 15–20 cm accounted for the lowest contribution (19%) to AGB in the study area, while the 10–15 cm strata accounted for the highest AGB contribution (32%).

Aboveground Biomass

We estimated a mean plot AGB of 83.7 Mg ha^{-1} (±63) ranging from 11.1 to 241.2 Mg ha^{-1} (**Supplementary Table 1**). Plots with the highest biomass were found in the community protected site plots in Ete site, and Kono located close to the mouth of the Imo estuary. The lowest biomass was observed in the inland creek (Oproama) sites where shrub red mangroves (*Rhizophora mangle*) were dominant and urbanization actively taking place (**Supplementary Table 1**). The highest AGB (241 Mg ha^{-1}) in the study was observed in the undisturbed plot in Ete (E3A) where mangroves wetlands were protected from logging. In contrast, the lowest AGB (11 Mg ha^{-1}) was found in the disturbed plot in Oproama (O1C) community where a power line was constructed to bring electricity to the community. There was a significant difference between the two river estuaries (Sombreiro and Imo) of the study area following a one-way ANOVA. SRE had a significantly higher AGB compared to IRE ($p < 0.001$, mean difference = 69 Mg ha^{-1}).

We observed that stem density had a significant positive correlation with AGB [$p \leq 0.00001$, Spearman's rho (r_s) = 0.88], thus the higher the stem density, the higher the AGB in the plots. Results of the Spearman's correlation indicated there was a significant positive relationship ($p < 0.05$, $r_s = 0.47$) between AGB and distance from closest settlement. The farther the plots were from the settlement, the higher the AGB in the region. There was also a significant negative correlation between AGB and distance from the sea ($p < 0.05$, $r_s = -0.41$) and tidal channel ($p < 0.05$, $r_s = -0.50$).

Leaf Area Index

Leaf area index across the study ranged from 0.08 to 2.78, with a mean of 1.45. The highest plot mean LAI (2.41) was recorded in Ete (E1C), which is a heavily exploited site with nipa palm (*Nypa fruticans*) colonization. We recorded the lowest plot mean (0.42) in Oproama (O1C) another heavily exploited site with less dense vegetation. We observed a significant difference in LAI between the SRE and IRE. *Post hoc* comparisons using Tukey test indicated that IRE had a significantly higher mean LAI of 0.64 compared to SRE. We also analyzed the LAI measurements variance within each plot to determine the data spread. We found the highest variation (1.17) in E1A at Ete, a heavily exploited site, which can account for irregular canopy cover. We found the lowest variation (0.01) in the protected sites Kono. However, there was no significant difference in LAI variance between the two river estuaries.

There was a significant positive correlation between LAI and AGB ($p < 0.01$, $r_s = 0.62$), stem density ($p < 0.001$, $r_s = 0.63$) and basal area ($p < 0.01$, $r_s = 0.60$) at plot scale ($n = 25$). Linear regression models, performed on 60% of plot data, indicated that 31% of LAI accounted for plot AGB ($R^2 = 0.31$, $p < 0.001$) (**Figure 3**). The root mean square error (RMSE) was 59 Mg ha^{-1} from the calibration data while independent validation had an RMSE of 48 Mg ha^{-1}.

Disturbance Regime

The process of classifying plots into different disturbance regimes is complicated as various factors could contribute to the measured and observed criteria for characterization. For example, the presence of nipa was only prominent in the seaward IRE (Kono and Ete). The second complication is the mode of exploitation of different plots depending on location. Mode of exploitation ranged from logging for fuelwood (IRE: Ete), total clearance for construction activities (SRE: Oproama) and passage for boat transport (all locations). Disturbed plots in the Ete site were primarily affected by stem cutting for fuel, which is the primary source of income in the region. The nipa palm is now gradually invading these plots (**Supplementary Figure 5A**). We surveyed fourteen (14) disturbed and eleven (11) undisturbed plots (**Supplementary Table 1**). The resulting exploited plots

TABLE 1 | Proportional contribution of different diameter at breast height (DBH) size classes to stem density, basal area and aboveground biomass (AGB) of the study region.

Class	DBH range (cm)	Stem density (trees ha^{-1})	Percentage contribution of stem density (%)	Basal area (m^2 ha^{-1})	Basal area percentage contribution (%)	Aboveground biomass (t ha^{-1})	AGB percentage contribution (%)
1	5.0 to <10.0	588	65.0	2.82	34.4	21.2	27.0
2	10.0 to <15.0	239	26.0	2.74	33.5	25.0	32.0
3	15.0 to <20.0	57	6.0	1.35	16.5	14.5	19.0
4	≥20	24	3.0	1.27	15.6	16.8	22.0

in Oproama site was primarily due to a historical disturbance. About 6 ha of mangrove forest was cleared in 2013 to create a path for the construction of power lines (**Supplementary Figure 5B**). The high level of shell fishing was also evident in these plots where mangrove was cleared in order to make more waterways to the mangrove interiors, hence reducing travel time to fishing site. The plots in Kono are located adjacent to a protected mangrove site but there was presence of nipa along the fringes of mangrove forests (**Supplementary Figure 3E**). The locals culturally protect the Kono plots by manually removing nipa seedlings from the mangrove forest floor. We observed numerous nipa seedlings on the forest floor of this site (**Supplementary Figure 5C**).

Disturbance Regime, Aboveground Biomass, and Leaf Area Index

We observed significant difference in plot structural characteristics between disturbance regimes using a one-way ANOVA. Undisturbed plots had a significantly higher AGB ($p < 0.0001$) and LAI ($p < 0.01$) than disturbed plots. There was no significant difference in LAI variation between the disturbance regimes. A two way ANOVA was carried out on plot AGB and LAI variation independently by disturbance and river estuary. There was no significant interaction between the effects of disturbance regime and estuary on AGB and LAI variance.

Disturbance Regime and Stand Structure

We observed significantly different plot stem density amongst disturbance regime using a one way ANOVA. Tukey's HSD *post hoc* tests indicated that disturbed plots had significantly lower stem density than undisturbed plots (mean difference = 1,158 stem ha^{-1}, $p < 0.001$). A one way ANOVA was conducted to check the significant contribution of DBH size classes contribution to plot density, basal area and AGB among disturbance regime. There was significantly higher contribution of DBH size class 15–20 cm to plot density [$F(1, 23) = 2.428$, $p < 0.05$, mean difference = 6%], basal area [$F(1, 23) = 1.426$, $p < 0.05$, mean difference = 8%] and AGB [$F(1, 23) = 365.3$, $p < 0.05$, mean difference = 8%] in undisturbed plots compared to disturbed plots. There was significantly higher contribution of DBH size class 10–15 cm to basal area [$F(1, 23) = 312.5$, $p < 0.05$, mean difference = 7%] in disturbed plots compared to undisturbed plots. In the undisturbed regime, the highest DBH size class (>20 cm) made up about 3% of the mean plot stem density, but contributed 24% of the AGB. The percentage contribution of each DBH size class to the AGB in the undisturbed regime were more evenly distributed (20–30%) compared to the HE and ME regimes, where the lowest two DBH size classes 5–10 cm and 10–15 cm made up about 70% of the AGB (**Figure 4** and **Table 2**).

Nipa Stand Patterns

We recorded 179 (0–33) nipa palm stands during the survey (**Supplementary Table 1**). We observed no nipa palm colonization in the inland Oproama plots (SRE) but recorded nipa palm invasion in seaward Ete and Kono (IRE). The absence of nipa palm in SRE is because it is further inland from the point of introduction. We recorded five plots with high nipa invasion (**HI**), seven plots with moderate invasion (**MI**) and two plots with no nipa (**NI**) stand during the field study in IRE (**Supplementary Table 1**). We excluded plots in SRE from further analysis because there was no sign of nipa invasion. Analysis of nipa population in IRE showed significant negative correlation with mangrove basal area ($p < 0.01$; $r_s = -0.74$), AGB ($p < 0.01$; $r_s = -0.77$), stem density ($p < 0.05$; $r_s = -0.57$). There was no correlation with LAI. However, nipa population had a significant positive correlation with plot LAI variation ($p < 0.05$; $r_s = 0.71$) and proportion of basal removed ($p < 0.05$; $r_s = 0.73$). Nipa stand density showed a negative correlation to the contribution of DBH size class 2 (10–15 cm) to basal area ($p < 0.05$; $r_s = -0.45$) and AGB ($p < 0.05$; $r_s = -0.45$). There was significantly higher nipa stands in disturbed plots than undisturbed plots following a one-way ANOVA ($p < 0.001$). We also performed one way ANOVA of LAI variance and proportion of basal area removed individually amongst the level of nipa invasion. No invasion plots showed significantly lower LAI variation to MI ($p < 0.05$, mean difference = 0.6) and HI plots ($p < 0.01$, mean difference = 0.8). Heavily invaded plots also showed significantly higher proportion of BA removed to MI ($p < 0.01$, mean difference = 40%) and NI ($p < 0.01$, mean difference = 60%) plots.

DISCUSSION

We performed the most comprehensive mangrove stem, biomass and canopy structure survey in Nigeria, the largest mangrove forest in Africa. We showed a general pattern of AGB across gradients from sea, tidal channel and closest settlement. We also showed the degree of disturbance (wood exploitation and clearance for development) affecting AGB in the region. We also addressed the subtle effect of local disturbance on stem size distribution, the possible target size class by locals and the resultant invasion of nipa palm (*Nypa fruticans*) in mangrove forests of the Niger Delta. We provided evidence of the relationship between mangrove clearing and the invasion of nipa palm. This is the first report in West Africa to give evidence of the relationship between stand structure and invasive species colonization in mangrove forests.

Aboveground Biomass and Leaf Area Index Patterns in the Niger Delta

Forest productivity has a direct influence from nutrient availability and external influence by anthropogenic disturbance (Alongi, 2009). Here, we report the relationship between AGB and tidal influence of mangrove forests in the Niger Delta. We observed that higher AGB and BA were located in plots with closer proximity to the ocean and tidal channel. This could be because of the influence of nutrient mixing effect of tide on mangrove forests (Harris et al., 2010; Carugati et al., 2018). Castaneda (2010) reported that mangrove productivity in South Florida mangrove forests may be limited by phosphorus fertility which showed a negative gradient with distance from the ocean. He also gave evidence that tidal inundation duration and frequency influences the fertility of mangrove soils, hence

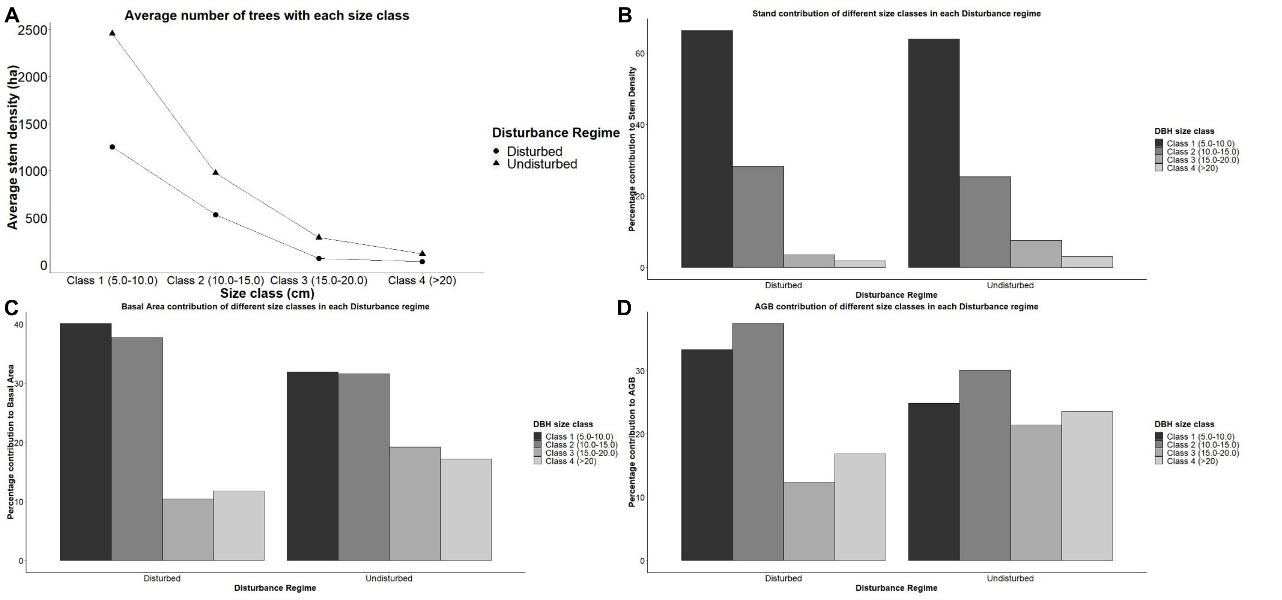

FIGURE 4 | Percentage contribution of size classes to within plots of the same disturbance group. **(A)** Average number of trees with each size class. **(B)** Stand contribution of different size classes in each Disturbance regime. **(C)** Basal Area contribution of different size classes in each Disturbance regime. **(D)** AGB contribution of different size classes in each Disturbance regime.

TABLE 2 | Proportion of diameter at breast height (DBH) size classes contributing to stand density, basal area, and aboveground biomass (AGB) in different disturbance regime.

Disturbance regime	DBH size class (cm)	Number of stems	Stand density (stem ha^{-1})	Stand density proportion (%)	Basal area (m^2 ha^{-1})	Basal area proportion (%)	AGB (t ha^{-1})	AGB proportion (%)	LAI mean (variance)
Disturbed (3.5 ha)	Class 1 (5.0–10.0)	1,252	329	66.3	1.7	40.1	12.5	33.4	1.12 (0.45)
	Class 2 (10.0–15.0)	533	140	28.2	1.6	37.7	14.1	37.5	
	Class 3 (15.0–20.0)	69	18	3.7	0.4	10.4	4.6	12.3	
	Class 4 (>20)	35	9	1.9	0.5	11.7	6.3	16.9	
Undisturbed (2.25 ha)	Class 1 (5.0–10.0)	2,457	980	64.0	4.6	31.9	34.4	24.9	1.87 (0.25)
	Class 2 (10.0–15.0)	976	389	25.4	4.5	31.7	41.6	30.1	
	Class 3 (15.0–20.0)	291	116	7.6	2.8	19.2	29.6	21.4	
	Class 4 (>20)	116	46	3.0	2.5	17.2	32.6	23.6	

productivity (Castañeda-Moya et al., 2013). Although there has been no reports of mangrove AGB and salinity in the region, Ukpong (1991) reported that *Rhizophora racemosa* (salinity range: 1.7–5.9%) was more adapted to saline conditions than *R. mangle* (salinity range: 1.4–4.4%). The significant difference in mangrove forest structure and biomass across the tidal gradient during this study may be linked to this influence. Soil nutrient and salinity variation explained the main relationship between mangrove vegetation and soil in the western Niger Delta (Ukpong, 1994). Sherman et al. (2003) reported a trend in AGB across tidal gradient being a function of salinity stress. However, like our study, he showed that more inland mangrove forests had lower stands compared to the seaward forests. Further affecting the biomass pattern of mangrove forests in the Niger Delta could be the proximity to settlements, which may alter the productivity of mangrove forests through anthropogenic disturbance and nutrient pollution (Lewis et al., 2011; Maiti and Chowdhury, 2013; Scales and Friess, 2019). Mangrove productivity patterns can be altered by pollution in the Niger Delta, evident from increased litter fall (Numbere and Camilo, 2018). Although, some studies have reported that seaward distance has more effect on soil organic carbon (SOC) than AGB (de Jong Cleyndert et al., 2020). Tidal flooding which maybe as a result of heavy rainfall may contribute to the phosphorus balance of mangrove forests

(Chen and Twilley, 1999). Our study showed that AGB were lower at plots closer to settlements, which could be as a result of either nutrient modification, pollution, or a consequence of perturbation through logging and fishing.

Our field measurements of LAI (0.08–2.78) are within range of LAI measured by hemispherical photograph (Wong and Fung, 2013; Prananda et al., 2020). The recorded LAI in this study are also lower compared to alternative field LAI measuring methods such as Plant Canopy Analyzer (Clough et al., 1997; Kamal et al., 2016). Our low records of LAI (<3) could be because of the stage of the mangrove forests, disturbance and methodology. We employed the indirect method for estimating LAI in this study using hemispherical photography, which has been seldom employed in the estimation of LAI in mangrove forests. This method has been known to underestimate LAI and could explain the high difference in LAI from comparative studies using direct methods (Ishil and Tateda, 2004). Hence, multiple measuring techniques should be used in comparison for mangrove canopy structure. Low recorded LAI in our study could also be a representation of the state of the mangrove forests in the Niger Delta. A study by Pool (1973), showed that mangrove forests in early succession have reduced LAI while higher LAI can be characteristic of later succession, especially in mixed stands (Pool, 1973). This was evident in this study where we recorded the lowest LAI in a site that was cleared in 2013 (Oproama) for a power line construction. This low LAI in some of the plots could be as a result of the sites being in early succession as a result of a previous disturbance (Pool, 1973). Another reason for the low values of LAI in this study could be the monospecific *Rhizophora* nature of the sample plots. Clough et al. (1997) characterized *Rhizophora* spp. as canopy shy (shade intolerant) because of the numerous light gaps between trees, which could increase the area of light penetration within plots. Higher LAI have also been recorded heterogeneous forests compared to homogeneous forests in Australia (Kamal et al., 2016). The relationship between LAI and the state of mangrove forests creates an opportunity to map mangrove productivity in Niger Delta from a relationship between LAI and vegetative indices of earth observation satellites (Kovacs et al., 2009; Manna and Raychaudhuri, 2020). Monitoring landscape productivity is a vital means of assessing rate of deforestation in mangrove forests. LAI is a proxy for primary production through its relation to photosynthetic capacity of the canopy (Pool, 1973; Araujo et al., 1997; Williams et al., 1997). We showed a significant positive correlation (r_s = 0.63) and a regression equation (R^2 = 0.39) between LAI and AGB indicating the potential for LAI to be used as a proxy for mangrove productivity in the Niger Delta.

Local Disturbance Effect on Biomass, Stand Structure, and Canopy Properties

Stand structure in a forest landscape provides a means to monitor the effects of disturbances on forests ecosystems. Natural sources of perturbation such as hurricane and anthropogenic sources such as wood harvesting can modify the stand structure of the ecosystem thereby affecting the productivity of mangrove forests (Wan Norilani et al., 2014). The DBH size class in our study showed a reverse J-shaped distribution, which has been reported by other mangrove stand reports (Kuei-Chu, 2008; Nguyen et al., 2020). Nguyen et al. (2020) reported this reverse J-shaped distribution in erosion sites, while Kuei-Chu (2008) gave this as an evidence of regeneration. Despite showing this reverse J-shaped distribution, indicative of disturbance or regeneration, our study excluded DBH <5 cm. Hence, we could not make further inference of this distribution. Further study can include DBH <5 cm and include evidence of regeneration in plots. We observed that DBH size class contribution to AGB had a significant difference between disturbance regimes with disturbed plots having an uneven contribution to AGB. Wan Norilani et al. (2014) reported that there was a uniform distribution of stem size classes in a naturally disturbed area compared with a harvested region at Kisap Forest Reserve, Malaysia (Wan Norilani et al., 2014). This uniformity was also reflected in this research where the contribution of each stem size class to the AGB in undisturbed plots was more even (20–30%) than exploited plots, where the lowest two classes (5–10 cm and 10–15 cm) made up about 70% of the AGB. Symmetrically distributed diameter of mangrove stands have also been observed in Malaysia (Jusoh and Aziz, 2014). Wah et al. (2011) reported that disturbed mangrove stands in Semporna, Malaysia had lower DBH range (10–20 cm) and less dense stands compared to undisturbed plots (20–35 cm). Natural disturbance can alter stand structure of mangrove forests (Castañeda-Moya et al., 2010; Biswas et al., 2012; Cox et al., 2016), however, our results have shown that anthropogenic disturbance from wood extraction for fuel wood and development can also alter the stand properties of mangrove forests in the Niger Delta.

The stem size class 3 (15–20 cm) had the lowest contribution to AGB in all disturbance groups. This has an implication on the target stem size for harvest. This stem size class is the target tree size within the region. Scales and Friess (2019) also indicated a preferential selection of DBH >10 cm of *Rhizophora mucronata* trees in the Bay of Assassins, Madagascar. The target tree size is the most convenient tree size to harvest in order to maximize the effort of loggers by reducing the cost of logging and transportation in order to increase profit. Another possible reason for the targeted DBH size class is the location of the larger stem size class close to the tidal channel would have resulted in harvested wood falling into the creek. The implication of selectively harvesting mangrove stand is the resulting shift in forest stand structure (Scales and Friess, 2019). Walters (2005a) also reported the species and size selectivity of mangrove stands based on its utilization. He reported that *Rhizophora* spp. was primarily cut for local use in fishing and fuel wood due to its slow decay property. Selective cutting practices could also be used to conserve older trees which contribute to larger carbon pools (Rasquinha and Mishra, 2020). The effect of target harvesting of mangrove stands results in the change in stand structure and light gap creation within mangrove stands (Clarke and Kerrigan, 2000; Duke, 2001; Amir and Duke, 2009; Mohamed et al., 2009).

The incidence of cutting mangroves in the Niger Delta could be because of a shift in economy. There was a historical reason for the cutting of mangroves as a source of income due to a shift from fishing primarily because of bad fishing practice, including chemical harvesting and use of small mesh sizes (Personal Communication, 2017). The use of Gamallin-20 (a paralyzing fish chemical) and small net mesh sizes resulted in depleting fish stock in the region (Olaoye and Ojebiyi, 2018). This shift in economy and the dependence of local communities on mangrove stands for fuel wood puts a further pressure on the mangrove forests in the Niger Delta. A similar dependence on mangrove stands by local communities in Southern Cameroun also caused damage to mangrove stands and environmental changes in mangrove forests (Nfotabong-Atheull et al., 2011). Mangrove stem is one of the major sources of income in local communities in the Niger Delta, who also use it for fuel wood. Wood harvesting practices can be employed to create sustainable management of mangrove forest resources.

The change in stand structure of mangrove forests as a result of wood exploitation also has an effect on canopy properties, hence, modifying LAI (Araujo et al., 1997). Anthropogenic disturbance within the study region resulted in LAI variation where exploited plots had a higher variation in LAI than undisturbed plots. Light gaps in mangrove forests are naturally created from dead mangrove trunks caused by hurricane, lightening and diseases (Amir and Duke, 2009), but these can also be created by small-scale disturbance through wood extraction or clearance (Duke, 2001). Irregular harvesting, as seen in exploited plots in the study resulted in open gaps within these plots. These light gaps have an implication on regeneration and recruitment of mangrove trees (Duke, 2001; Mohamed et al., 2009). We observed regeneration in some of these plots within forest gaps in disturbed plots during field surveys. However, due to the invasion of nipa palm, the regeneration of mangrove stands in light gaps within the study region is hindered. We observed young mangrove and nipa stands growing in forest gaps, competing for forest resources. This is a common feature along the Imo estuary (Ete and Kono plots). However, nipa palm always outcompetes *Rhizophora* spp. (Numbere, 2019). The effect of selective harvesting can have a negative influence in the natural growth of a mangrove forest especially the presence of an invasive species to colonize available cleared mangrove area in the Niger Delta.

Pattern of Nipa Palm Invasion

There is a growing interest of invasion ecology globally due to its influence on ecosystem function and economic impacts (Secretariat of the Convention on Biologival Diversity, 2010). Ukpong (2015) has reported on *Nypa fruticans* zonation and soil conditions in Niger Delta mangrove forests. We reported here a possible cause of nipa palm (*Nypa fruticans*) colonization in Niger Delta mangrove forest. Reports have shown the close link between mangrove deforestation and non-native species colonization (Harun Rashid et al., 2009). Harun Rashid et al. (2009) gave evidence that the colonization of non-native invasive species in Bangladesh can be as a result of forest gaps formed from catastrophic events affecting mangrove forests (Harun Rashid et al., 2009). We reported a similar trend in this study where higher LAI variance and proportion of BA removed were significantly higher in plots classified as high nipa invasion (**HI**) compared to moderate (**MI**) and no invaded plots (**NI**). This relationship is an indication that nipa seedlings slowly colonize light gaps within disturbed mangrove forests. Ukpong (2015) also argued that the slow development process of red mangrove (*Rhizophora* spp.) regeneration and its inability to regenerate after being cut also aids in nipa outcompeting these natural species. The poor ability of *Rhizophora* spp. to regenerate from cut stem or from the root has been reported from various studies (Hamilton and Snedaker, 1984; Food and Agriculture Organization, 1994; Walters, 2005a,b). Hamilton and Snedaker (1984) reported that *Rhizophora mangle* cannot regenerate when cut from the roots, stump, and trunk but could grow back from loss of foliage of about 50% and cut branches more than 2 cm diameter. Numbere (2019) also gave evidence that nipa palm had a competitive advantage to mangrove based on soil properties, hence, further research can investigate the relationship among nipa invasion, mangrove recruitment and soil properties. A case can also be made for the inclusion of nipa palm in biomass estimates of mangrove forests in coastal Nigeria.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author.

AUTHOR CONTRIBUTIONS

CN conceived the research and carried out the field measurements and statistical analyses. CN and MW designed the work, contributed to interpretation of results, and wrote the manuscript. Both authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/ffgc.2021.746671/full#supplementary-material

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Brazilian Mangroves: Blue Carbon Hotspots of National and Global Relevance to Natural Climate Solutions

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Mangroves are known for large carbon stocks and high sequestration rates in biomass and soils, making these intertidal wetlands a cost-effective strategy for some nations to compensate for a portion of their carbon dioxide (CO_2) emissions. However, few countries have the national-level inventories required to support the inclusion of mangroves into national carbon credit markets. This is the case for Brazil, home of the second largest mangrove area in the world but lacking an integrated mangrove carbon inventory that captures the diversity of coastline types and climatic zones in which mangroves are present. Here we reviewed published datasets to derive the first integrated assessment of carbon stocks, carbon sequestration rates and potential CO_{2eq} emissions across Brazilian mangroves. We found that Brazilian mangroves hold 8.5% of the global mangrove carbon stocks (biomass and soils combined). When compared to other Brazilian vegetated biomes, mangroves store up to 4.3 times more carbon in the top meter of soil and are second in biomass carbon stocks only to the Amazon forest. Moreover, organic carbon sequestration rates in Brazilian mangroves soils are 15–30% higher than recent global estimates; and integrated over the country's area, they account for 13.5% of the carbon buried in world's mangroves annually. Carbon sequestration in Brazilian mangroves woody biomass is 10% of carbon accumulation in mangrove woody biomass globally. Our study identifies Brazilian mangroves as a major global blue carbon hotspot and suggest that their loss could potentially release substantial amounts of CO_2. This research provides a robust baseline for the consideration of mangroves into strategies to meet Brazil's intended Nationally Determined Contributions.

Keywords: Brazil, mangrove forests, blue carbon, hotspot, CO_2 equivalent emissions

INTRODUCTION

Climate change velocity has outpaced models' predictions spurring the implementation of natural climate solutions policies centered on ecosystems self-organizing properties to mitigate fossil fuels emissions and ensue adaptive capacity to future alterations in the climate system. Natural ecosystems have evolved mechanisms that allow them to shift among alternate states while remaining functional over geomorphic timescales (Holling, 1973). Such processes are evident in dynamic coastal sedimentary environments, which alternate between vegetated and unvegetated states (e.g., saltmarshes and mangroves versus mudflats and saltflats) in response to climate and millennial-scale changes in sea levels (Gabler et al., 2017; Saintilan et al., 2020). In particular, where sediment yield to coastal oceans has not been impaired and coastal floodplains still allows for inland expansion, rising sea levels can increase accommodation space along mangrove- and marsh-dominated environments sustaining continuous burial of terrigenous and marine organic sediments (Rogers et al., 2019).

Among tidal saline wetlands, mangroves are known for high rates of carbon sequestration in soils (mean = 222 gC m^{-2} yr^{-1}; Jennerjahn, 2020; MacKenzie et al., 2020; Wang et al., 2020), that are 50 times higher than reported for terrestrial tropical and temperate forested biomes (mean = 4.5 gC m^{-2} yr^{-1}; McLeod et al., 2011). Combined with comparable carbon sequestration rates in woody biomass (mean = 82.7 gC m^{-2} yr^{-1}, range = 13–2,160 gC m^{-2} yr^{-1}; Xiong et al., 2019), these intertidal wetlands can be a cost-effective strategy for some nations to compensate for part of their carbon dioxide (CO_2) emissions (Taillardat et al., 2018). To date, however, few countries have the country-level inventories required to support the inclusion of coastal wetlands into national carbon credit markets (e.g., Holmquist et al., 2018 for the United States and Serrano et al., 2019 for Australia). Moreover, global estimates generally focus on carbon stocks within either soil or biomass (Hutchison et al., 2014; Jardine and Siikamäki, 2014; Atwood et al., 2017; Rovai et al., 2018, 2021b; Sanderman et al., 2018; Tang et al., 2018; Simard et al., 2019; Kauffman et al., 2020), which are important to determine potential CO_{2eq} emissions from mangrove forest loss (see Adame et al., 2021), but do not provide comparable information in terms of mitigating current emission rates. Further, global estimates often do not accurately quantify within-country variability, relying, in many cases, on averaged reference values or model-based generalizations to extrapolate predictions to data-poor or data-absent nations when harnessing national datasets would be more appropriate to inform country-specific conservation targets (Worthington et al., 2020a).

Brazil is home to the second largest mangrove area in the world, with forests distributed across diverse coastal morphology and climate gradients (Hamilton and Casey, 2016; Worthington et al., 2020b). Despite accounting for over 9% of the world's mangroves, Brazil still lacks an integrated inventory of carbon stocks and carbon sequestration rates that capture the diversity of coastline types and climatic zones in which mangroves are present. To fill this gap, we performed a comprehensive review of published global datasets to derive within-country estimates of carbon stocks and sequestration rates in mangrove soils and biomass that represent both geographic gradients and administrative divisions in Brazil. In addition to delivering state-level estimates, we provide a direct comparison between mangroves and Brazil's other major vegetated biomes, identifying mangroves as a major carbon hotspot that can help meet intended Nationally Determined Contributions (NDC's), in addition to their significance as global coastal carbon sinks.

MATERIALS AND METHODS

Data Acquisition

Geospatial Datasets and Analyses: Carbon Stocks in Biomass and Soils

Global mangrove aboveground biomass (AGB) and soil organic carbon stock (SOC) values were retrieved from various independent datasets that have explicitly mapped the spatial distribution parameters' (**Table 1**). These global datasets were subsetted for Brazilian mangroves, and median statistics were computed from grided or vectorized datasets where available or directly from the original references. Where possible, uncertainties were assessed on the basis of bootstrapped 95% confidence intervals for medians using the bias corrected and accelerated (BCa) method (Carpenter and Bithell, 2000; Mangiafico, 2021).

As noted elsewhere (Bukoski et al., 2020), due to the scarcity of field observations there are no regional or global mangrove belowground biomass (BGB) maps. Thus, to be consistent with previous studies, we used a BGB:AGB ratio of 0.5 to estimate BGB across the world's mangroves (IPCC, 2014; Hamilton and Friess, 2018; Rovai et al., 2021b). Further, biomass (both AGB and BGB) was converted to carbon units using a conversion factor of 0.475 (Hamilton and Friess, 2018).

To warrant direct comparison among independent sources, we standardized per-area (MgC ha^{-1}) and total (TgC or PgC) carbon stock estimates across AGB and SOC datasets using a conservative mangrove extent of 82,849 and 7,675 km^2 for the world's and Brazilian mangroves, respectively (**Table 1**; after Hamilton and Casey, 2016 but see Hamilton et al., 2018; Worthington et al., 2020a for comprehensive discussions on existing mangrove extent databases).

Biomass (AGB and BGB) and SOC (top 1 meter) stock estimates for Brazilian mangroves used throughout this study were computed from Rovai et al. (2018, 2021b) respectively, given the comparatively larger number of observations (>900 forest plots for AGB and >65 sites for SOC stocks distributed only within Brazil's mangroves; **Supplementary Table 1**) used in these studies. It is noteworthy that mean AGB and SOC estimates for global and Brazilian mangroves are consistent to mean values computed among previous studies (**Table 1**). Biomass (AGB and BGB) and SOC (top 1 meter) density in other Brazilian vegetated biomes (Amazon forest, Atlantic forest, Pampa grasslands, Cerrado savannas, Pantanal wetlands, and Caatinga forests) were extracted from harmonized biomass (Spawn et al., 2020) and soil (Hiederer and Köchy, 2011) databases. Due to some overlap between spatial datasets, cells containing mangroves were excluded when computing biomass and SOC density estimates for other Brazilian vegetated biomes. Global datasets were clipped to Brazil's territory, split by state-level administrative divisions and classified into vegetated biomes

TABLE 1 | Published above- and belowground biomass (AGB and BGB), and soil organic carbon (SOC) stock estimates for global and Brazilian mangroves.

Source	Mean AGB ($MgC\ ha^{-1}$)		Mean BGB ($MgC\ ha^{-1}$)		Mean SOC ($MgC\ ha^{-1}$)	
	Global	Brazil	Global	Brazil	Global	Brazil
Rovai et al., 2018, 2021b	78	66	39	33	297[a]	241[a]
Kauffman et al., 2020	115[b]	125[b]	741[b]	347[b]	334	155
Simard et al., 2019	58	42	29	21	283[c]	
Hamilton and Friess, 2018	98		49			
Tang et al., 2018	69	78	42	31		
Sanderman et al., 2018					361	358
Atwood et al., 2017					283	308
Hutchison et al., 2014	87	80	34	30	447	
Jardine and Siikamäki, 2014					369	342
Overall mean	78 ± 7	67 ± 9	39 ± 3	29 ± 3	329 ± 16	281 ± 37

Source	Total AGB (PgC)		Total BGB (PgC)		Total SOC (PgC)		Ecosystem-level C (PgC)	
	Global	Brazil	Global	Brazil	Global	Brazil	Global	Brazil
Rovai et al., 2018, 2021b	0.81	0.06	0.41	0.03	2.26[a]	0.16[a]	3.48	0.25
Kauffman et al., 2020	0.95	0.05	2.90[b]	0.13[b]	2.70	0.12	6.55[b]	0.30[b]
Simard et al., 2019	0.46	0.03	0.23	0.02	2.14[c]		2.83	
Hamilton and Friess, 2018	0.8		0.41		2.96[d]		4.17	0.39
Tang et al., 2018	0.56	0.06	0.34	0.02				
Sanderman et al., 2018					3.80	0.27		
Atwood et al., 2017					2.60	0.24		
Hutchison et al., 2014	0.72	0.06	0.28	0.02	3.64		4.64	
Jardine and Siikamäki, 2014					2.96	0.26		
Overall mean	0.72 ± 0.073	0.05 ± 0.006	0.33 ± 0.035	0.02 ± 0.003	2.99 ± 0.25	0.21 ± 0.03	3.78 ± 0.40	0.32 ± 0.07
Brazil's % of global	6.9%		6.1%		7.0%		8.5%	

[a] *Based on Rovai et al. (2018).*
[b] *Not included in the overall mean computation since per unit area values were >30 and >90% higher than mean AGB and SOC values computed from all other studies.*
[c] *Based on Atwood et al. (2017); not included in the overall mean computation.*
[d] *Based on Jardine and Siikamäki (2014); not included in the overall mean computation.*

according to the Brazilian Geography and Statistics Institute databases (IBGE, 2019).

Literature Search: Carbon Sequestration in Biomass and Soils

Carbon sequestration in mangrove woody biomass and soils were estimated based on a comprehensive literature review performed online on Google Scholar, Science Direct, Web of Science, and the Brazilian SciELO databases. For carbon sequestration in woody biomass, we performed searches using the following expressions: "carbon sequestration," "carbon accumulation," "wood production," "biomass production," "stem growth," "basal area increment," and "DBH increment" always in combination with the terms "mangrove" and "Brazil." Altogether the searches returned a total of 1,000 articles (Google Scholar = 815, Science Direct = 51, and Web of Science = 134). For carbon sequestration in mangrove soils, we used the expressions "carbon sequestration," "carbon accumulation," "carbon burial," and "carbon accretion" again always in combination with the terms "mangrove" and "Brazil." Initial searches returned a total of 3,725 articles (Google Scholar = 3,240, Science Direct = 404, and Web of Science = 81). Searches performed at the Brazilian SciELO database included generic Portuguese terms "carbono" (for carbon) and "mangue*" (for mangrove or mangal), which returned a total of 19 studies. Only studies conducted in Brazilian mangroves that presented data on carbon sequestration in either woody AGB (N = 2) or soils (N = 7) were included in our analyses. Carbon sequestration rates in mangrove woody biomass and soils were classified into one of four coastal geomorphic types along Brazil's shoreline: deltas, estuaries, lagoons or open coasts (after Worthington et al., 2020b). Differences among those coastal typologies were assessed using analysis of variance for unbalanced designs (ANOVA function from R "car" package; Fox and Weisberg, 2019).

Carbon Dioxide Equivalents Emissions and Foregone Carbon Sequestration

Carbon dioxide equivalents (CO_{2eq}) for both carbon stock and carbon sequestration rate values were estimated using a CO_2:C stoichiometric ratio of 3.67 (i.e., $CO_2/C = 44/12 = 3.67$), which is used as a multiplying factor to convert carbon atoms to CO_2 molecules. Potential CO_{2eq} emissions were computed on a "stock-difference" basis (*sensu* Kauffman et al., 2017) using published mangrove biomass and soil carbon stock estimates (based on Rovai et al., 2018, 2021b as detailed above) and carbon sequestration rates (from the literature review). Further, we coupled degradation-specific carbon emission factors (after

Sasmito et al., 2019: Erosion AGB = 1, SOC = 1; Clearing AGB = 0.7, SOC = 0.21; Settlement AGB = 1, SOC = 0.66; Extreme weather AGB = 0.31, SOC = 0.14; Agriculture and aquaculture AGB = 0.83, SOC = 0.52) with a high-resolution map of drivers of mangrove forest loss (covering the period 2000–2016; after Goldberg et al., 2020) to determine the dominant historical cause of mangrove degradation for each Brazilian state. While some mangrove loss drivers may change over time, dominant degradation causes, particularly those driven by climate (e.g., erosion caused by sea level rise and extreme weather events, which affects 85% of the country's mangrove coverage; Goldberg et al., 2020), are likely to remain as a result of global climate change. Likewise, agriculture or aquaculture and clearing may be harder to reduce in Brazil in the years to come due to increasing relaxation of environmental regulations. Once determined, dominant state-level emission factors were multiplied by carbon stocks in AGB and in soils (top 1 meter) separately and then summed to compute ecosystem-level potential CO_{2eq} emissions for each mangrove cell in the gridded dataset (that is, AGB and SOC density estimates combined from Rovai et al., 2018, 2021b).

All raster and vector manipulations and geospatial analyses were performed using R (R Core Team, 2020) packages 'geobr' (Pereira and Goncalves, 2021), 'raster' (Hijmans, 2020), and 'rgdal' (Bivand et al., 2020).

RESULTS AND DISCUSSION

Carbon Stocks in Biomass and Soils

Based on recent global estimates (**Table 1**), Brazil holds on average 8.5% (or 0.32 PgC) of the world's mangrove organic carbon stocks, partitioned among AGB (0.05 PgC or 6.9% of global stocks), BGB (0.02 PgC or 6.1% of global stocks) and soils (0.21 PgC or 7.0% of global stocks). On a per-area basis, Brazilian mangroves store on average 66, 33, and 241 MgC ha^{-1} in AGB, BGB and soils, respectively (from Rovai et al., 2018, 2021b for AGB and BGB, and SOC, respectively). Standardized to the same mangrove forest coverage, these values are comparable to and often more conservative than other studies' estimates. However, our ecosystem-level carbon stock estimate for Brazilian mangroves is 36% lower than that reported in Hamilton and Friess (2018) due to overestimated SOC density estimates for Brazil (from Jardine and Siikamäki, 2014) used in that study.

Over 80% of all mangrove carbon stocks in Brazil are found in the states of Maranhão (91.3 TgC), Pará (61.2 TgC) and Amapá (47.3 TgC), reflecting extensive coverage which amounts to more than 80% of the country's total mangrove area (**Table 2**).

Largest per-area AGB values are also found in these three states (215.5, 205.3, and 166.7 Mg ha^{-1} in Amapá, Pará and Maranhão, respectively) as well as in Piauí (143.4 Mg ha^{-1}), where mangroves develop in nutrient-rich deltaic systems. In contrast, lowest per-area AGB was found in Santa Catarina (56.8 Mg ha^{-1}), near the austral distribution limit for mangrove forests in the Southwestern Atlantic (Schaeffer-Novelli et al., 1990; Soares et al., 2012). AGB was also lower in São Paulo (84.2 Mg ha^{-1}) and Rio de Janeiro (83.1 Mg ha^{-1}), where extensive mangrove areas have been impacted by industrial activities and urban expansion (Soares, 1999; Ferreira and Lacerda, 2016; Moschetto et al., 2021). AGB values $<$100 Mg ha^{-1} were also found in Paraíba, Sergipe, Pernambuco, Ceará and Alagoas where shrimp farming has compromised the structural and functional integrity of Brazil's drier-climate mangrove forests (Lacerda et al., 2021). AGB values $>$100 Mg ha^{-1} were found in Espírito Santo and Bahia mangroves where the multidecadal stability of more than 70% of the mangrove coverage suggests that the integrity of

TABLE 2 | Median (95% Confidence Intervals) and total values for above- and belowground biomass (AGB and BGB) and, soil organic carbon (SOC) stock estimates for Brazilian states.

State	Mangrove area (ha)[a]	AGB (Mg ha^{-1})	SOC (Mg ha^{-1})	Total OC in AGB (Tg)	Total OC in BGB (Tg)[b]	Total SOC (Tg)	Ecosystem-level C (Tg)	Ecosystem-level C (%)
Maranhão (MA)	297,158.47	167 (160–171)	178 (174–179)	24.74	12.37	54.15	91.26	36.56
Pará (PA)	186,977.44	205 (200–208)	196 (173–209)	18.17	9.08	33.94	61.19	24.52
Amapá (AP)	141,625.98	215 (200–227)	209 (138–209)	14.26	7.13	25.92	47.31	18.95
Bahia (BA)	46,460.39	106 (90–114)	278 (276–279)	2.53	1.27	12.90	16.70	6.69
Paraná (PR)	19,581.39	99 (92–108)	269 (260–269)	0.97	0.48	5.26	6.71	2.69
São Paulo (SP)	14,776.24	84 (76–88)	270 (269–272)	0.60	0.30	4.07	4.97	1.99
Sergipe (SE)	10,056.71	98 (87–121)	286 (283–286)	0.53	0.26	2.90	3.69	1.48
Pernambuco (PE)	8,821.82	99 (93–121)	281 (276–281)	0.44	0.22	2.47	3.13	1.25
Paraíba (PB)	8,579.79	80 (75–84)	269 (268–269)	0.33	0.16	2.33	2.82	1.13
Rio de Janeiro (RJ)	7,182.39	83 (77–87)	293 (289–306)	0.35	0.17	2.21	2.73	1.09
Santa Catarina (SC)	6,430.90	57 (44–66)	285 (279–297)	0.21	0.10	1.82	2.14	0.86
Espírito Santo (ES)	5,796.23	119 (102–128)	292 (256–304)	0.29	0.14	1.68	2.11	0.85
Rio Grande do Norte (RN)	5,012.71	102 (93–105)	272 (268–272)	0.27	0.13	1.37	1.77	0.71
Ceará (CE)	3,532.48	79 (74–93)	253 (247–253)	0.16	0.08	0.89	1.14	0.46
Alagoas (AL)	2,826.20	97 (88–106)	284 (281–285)	0.13	0.06	0.81	1.00	0.40
Piauí (PI)	2,680.41	144 (80–182)	239 (237–239)	0.18	0.09	0.65	0.92	0.37
	Total			64.14	32.07	153.37	249.58	100

[a] *Estimated using Hamilton and Casey (2016) mangrove cover dataset.*
[b] *Estimated using Hamilton and Friess (2018) 0.5 AGB to BGB conversion factor.*
OC, organic carbon.

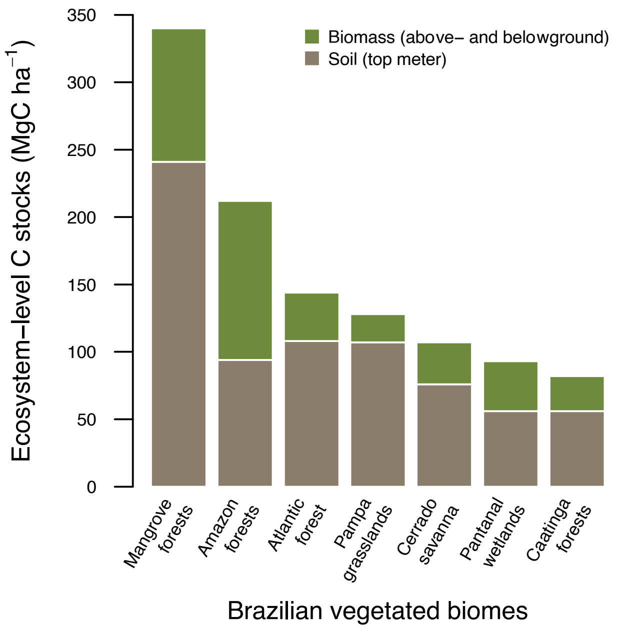

FIGURE 1 | Comparison of ecosystem-level carbon stocks (above-, belowground biomass and soil organic carbon in the top 1 meter combined) among major Brazilian vegetated biomes.

core areas have been maintained over time (Diniz et al., 2019). Predicted median AGB for Rio Grande do Norte mangroves was also >100 Mg ha^{-1} despite mangroves developing in a semi-arid climate and historical damage from shrimp farming (Lacerda et al., 2021). However, this result is likely due to the small number of observations used to constrain biomass predictions for that region (only two AGB values available for Rio Grande do Norte at the time Rovai et al., 2021b study was conducted; **Supplementary Table 1**). Regarding SOC stocks, deltaic mangroves in Piauí, Amapá, Pará and Maranhão states had lower soil carbon density due to higher inorganic-to-organic ratio per soil volume characteristic of coastal deltaic floodplains when compared to predominantly estuarine or lagoonal mangroves (Rovai et al., 2018; Sanderman et al., 2018; Jennerjahn, 2020; MacKenzie et al., 2020) found in other Brazilian states (**Table 2**). When summed, carbon stocks in biomass (AGB+BGB) and soils across Brazilian mangroves averaged 341 MgC ha^{-1} (range: 297–397 MgC ha^{-1}), showing little variation among states (e.g., maximum difference of 23% or ~80 MgC ha^{-1}) (**Table 2**). This relatively small variability in per-unit area carbon stocks reflect mangrove plants' resource partitioning strategies in response to broad geographical gradients (Rovai et al., 2021b), chiefly the role of coastal geomorphology in controlling the ratio between inorganic and organic matter in mangrove soils (Twilley et al., 2018; Jennerjahn, 2020).

Comparatively, on a per-area basis mangroves store between 2.2 and 4.3 times more carbon in the top meter of soil relative to other Brazilian vegetated biomes (**Figure 1**). Regarding mean carbon stocks in biomass (AGB and BGB combined), mangroves are second only to the Amazon forest, and 2.7–4.7 times higher than other Brazilian vegetation formations.

Carbon Sequestration in AGB and Soils

Currently, only two studies in Brazil report on carbon sequestration in mangrove woody AGB (**Table 3**). From these studies, carbon sequestration in Brazilian mangroves' woody AGB was estimated at 3.18 MgC ha^{-1} yr^{-1}, consistent with values reported for a diversity of coastal typologies worldwide (**Table 3**). Thus, we used this reference value to produce a first order country-level estimate of annual carbon sequestration in Brazilian mangrove AGB, which totals 2.44 TgC yr^{-1}, equivalent to 10% of all carbon sequestered in mangroves AGB globally.

Long-term carbon sequestration rates (mostly ^{210}Pb-dated cores) in Brazilian mangrove soils was estimated at 2.81 MgC ha^{-1} yr^{-1} (**Table 4**). While there were no differences (P > 0.05, results not shown) across the coastal geomorphic types found along Brazil's shoreline, this value is 15–30% higher than recent global estimates (e.g,. 1.94–2.39 MgC ha^{-1} yr^{-1}; Jennerjahn, 2020; MacKenzie et al., 2020; Wang et al., 2020), likely due to the predominance of minerogenic coastlines (deltaic, which accounts for >80% of the country's mangrove area, and meso- and macrotidal estuarine systems) where deposition of both autochthonous (mangrove detritus) and allochthonous (terrestrial and marine detritus) sediments are amplified (Adame et al., 2010; Kusumaningtyas et al., 2019; Cragg et al., 2020). Importantly, when this national median value is multiplied by the country's mangrove area coverage, annual carbon sequestration in Brazilian mangroves soils was estimated at 2.14 TgC yr^{-1}, corresponding to about 13.5% of the total amount of carbon buried annually in the world's mangroves.

Potential CO_{2eq} Emissions and Foregone Carbon Sequestration

Highest potential CO_{2eq} emissions (>900 MgCO_{2eq} ha^{-1}) resulting from loss of existing mangrove forests were estimated for Rio de Janeiro, Alagoas, Piauí, Pará, Amapá, and Maranhão states driven by the dominance of erosion (**Figure 2** and **Table 5**) where eventually all carbon stored in soils (here based on top 1 meter) and in AGB is lost to the atmosphere. It should be noted, however, that while eroded SOC is rapidly mineralized in aerobic estuarine waters (Sapkota and White, 2021), carbon release back to the atmosphere from biomass loss is not immediate given slow decomposition rates of downed wood in mangrove forests (Romero et al., 2005). Notably, when considering only the top 1 meter of soil to compute such estimates, these values are amongst the highest CO_{2eq} emissions reported in the literature for other mangrove sites worldwide (Kauffman et al., 2017; Alongi, 2020; Adame et al., 2021). Further agriculture/aquaculture- and settlement-based losses (emission factors of 0.83 and 1.00 for AGB and 0.52 and 0.66 for SOC, respectively) were also anticipated to cause high potential CO_{2eq} emissions (>500 MgCO_{2eq} ha^{-1}) in Espírito Santo, Pernambuco, Rio Grande do Norte, São Paulo, and Santa Catarina states as these activities represent a considerable loss of both aboveground and soil compartments (**Figure 2**).

TABLE 3 | Carbon sequestration rates in woody biomass for Brazilian and global mangroves.

Region	State	Typology	Wood NPP (MgC ha^{-1} yr^{-1})	Source
Southeast	Rio de Janeiro (RJ)	Open coast	2.64 ± 1.03	Estrada et al., 2015[a]
			1.90 ± 1.00	
			2.39 ± 1.45	
	São Paulo (SP)	Lagoon	7.03 ± 1.30	Data from Rovai et al., 2021a[b]
			4.06 ± 1.16	
			3.71 ± 1.07	
	Overall median (95% Confidence Intervals) Brazil		**3.18 (2.14–4.84)**	
Global		Deltas	3.64 ± 0.30	Data from Xiong et al., 2019[b]
		Estuaries	2.96 ± 0.39	
		Lagoons	4.64 ± 1.32	
		Open coasts	4.14 ± 0.63	
	Overall median (95% Confidence Intervals) global		**3.89 (2.96–4.39)**	

[a]*Mean ± 1SD as reported in the original study.*
[b]*Mean ± 1SE.*
NPP, net primary productivity.

TABLE 4 | Carbon sequestration rates in soils for Brazilian mangroves.

Region	State	Site	Typology	Carbon sequestration rate (MgC ha^{-1} yr^{-1})	Dating method	Source
North	Pará (PA)	Ajuruteua	Delta	2.54		Wang et al., 2020
Northeast	Pernambuco (PE)	Tamandaré	Estuary	3.53	210Pb	Sanders et al., 2010b
		Tamandaré	Estuary	9.49	210Pb	Sanders et al., 2010b
	Bahia (BA)	Jaguaripe	Estuary	1.26 ± 0.14[a]	210Pb	Hatje et al., 2021
		Jaguaripe	Estuary	1.28 ± 0.03[a]	210Pb	Hatje et al., 2021
		Jaguaripe	Estuary	2.89 ± 0.09[a]	210Pb	Hatje et al., 2021
		Jaguaripe	Estuary	3.37 ± 0.07[a]	210Pb	Hatje et al., 2021
		Jaguaripe	Estuary	4.08 ± 0.04[a]	210Pb	Hatje et al., 2021
‘		Jaguaripe	Estuary	7.76 ± 1.28[a]	210Pb	Hatje et al., 2021
Southeast	Espírito Santo (ES)	Caieira Velha	Estuary	2.82		Wang et al., 2020
		Vitoria	Estuary	3.79		Wang et al., 2020
		Anchieta	Open coast	4.30	210Pb	Wang et al., 2020
	Rio de Janeiro (RJ)	Ilha Grande	Open coast	1.86	210Pb	Sanders et al., 2008
		Ilha Grande	Open coast	1.69	210Pb	Sanders et al., 2010c
		Guanabara	Estuary	2.76		Wang et al., 2020
		Guanabara	Estuary	2.93		Wang et al., 2020
		Sepetiba	Open coast	5.85		Wang et al., 2020
	São Paulo (SP)	Cananéia	Lagoon	2.80 ± 0.14[b]	137Cs	Sanders et al., 2014
		Cubatão	Lagoon	10.21 ± 0.93[b,c]	137Cs	Sanders et al., 2014
		Cananéia	Lagoon	1.92	210Pb	Sanders et al., 2010a
		Cananéia	Lagoon	2.34	210Pb	Sanders et al., 2010a
South	Paraná (PR)	Paranaguá	Estuary	1.68	210Pb	Sanders et al., 2010c
		Guaratuba	Estuary	3.37	210Pb	Sanders et al., 2010c
	Overall median (95% Confidence Intervals)			**2.81 (1.92–3.37)**		

[a]*Mean ± 1SE computed from different depths within same cores for each site.*
[b]*Mean ± 1SE computed from two sites.*
[c]*Impacted site, not used to compute median and 95% CI's.*

Lowest potential CO_{2eq} emissions were linked to episodic extreme weather events that have the potential to release smaller fractions on carbon stored in AGB (31%) and soils (14%) followed by clearing, which can remove substantial carbon stocks in aboveground (70%) and soil (21%) compartments. These estimates are conservative considering only carbon stored in AGB and soils (but not in BGB, since emission factors for this plant compartment have not been established yet) were used to compute potential CO_{2eq} emissions resulting from mangrove forest loss.

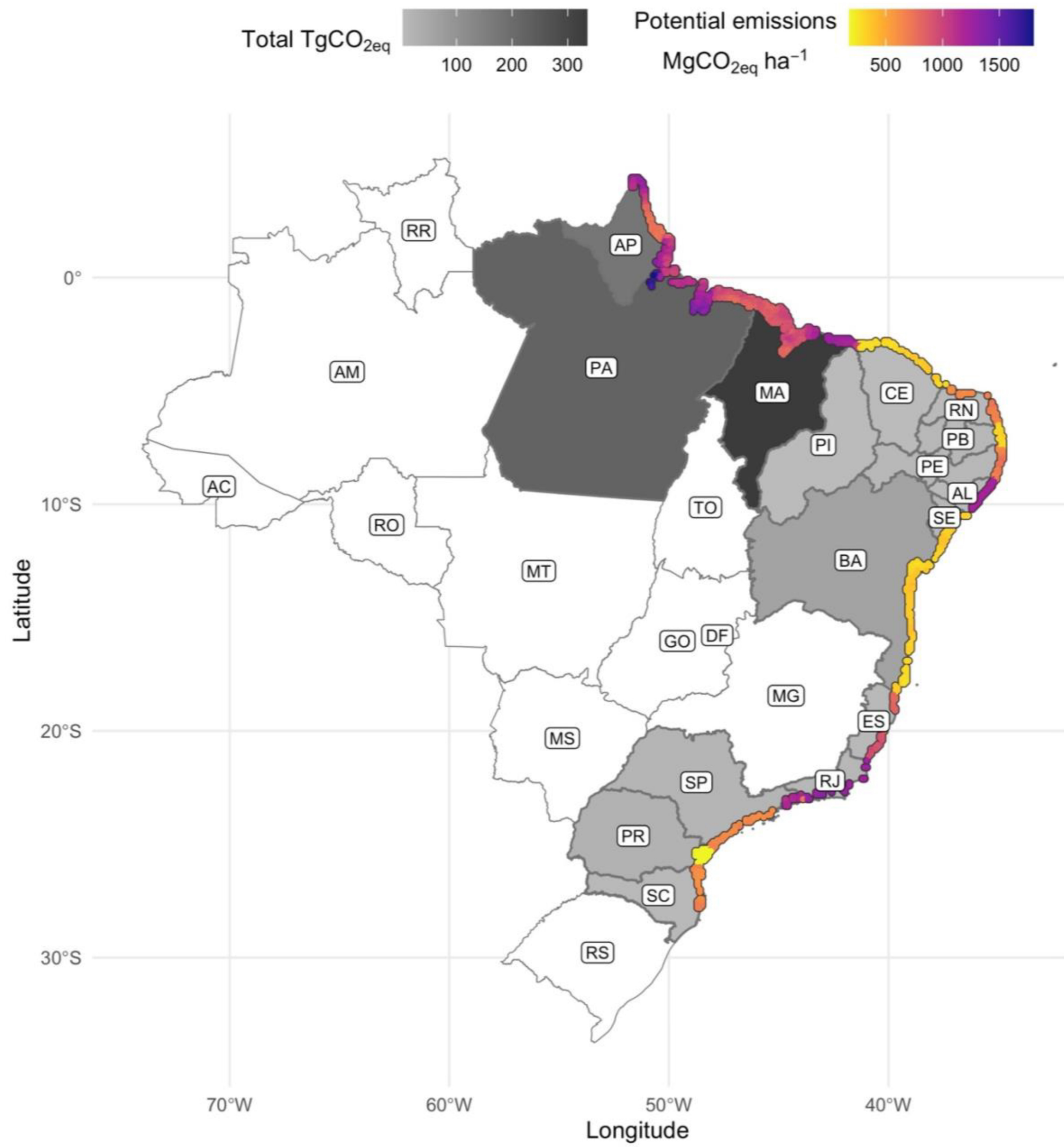

FIGURE 2 | Total CO_{2eq} stored in biomass and soils (grayscale top left legend) and variability in potential CO_{2eq} emissions (colored scale top right legend) across Brazilian mangroves. Mangrove coverage exaggerated to improve visualization. Estimates per state are given on **Table 5**. Mangrove states: AP, Amapá; PA, Pará; MA, Maranhão; PI, Piauí; CE, Ceará; RN, Rio Grande do Norte; PB, Paraíba; PE, Pernambuco; AL, Alagoas; SE, Sergipe; BA, Bahia; ES, Espírito Santo; RJ, Rio de Janeiro; SP, São Paulo; PR, Paraná; and SC, Santa Catarina. Non-mangrove states: RR, Roraima; AM, Amazonas; AC, Acre; RO, Rondônia; MT, Mato Grosso; TO, Tocantins; GO, Goiás; DF, Distrito Federal; MS, Mato Grosso do Sul; MG, Minas Gerais; RS, Rio Grande do Sul.

The loss of carbon sequestration potential after mangrove forests are degraded was assumed here to be 100% considering that soil and vegetation loss represent either acute stressors, which ceases mangrove production altogether, or chronic stressors that leave the system more susceptible to eventually collapse (Lugo et al., 1981; Lewis et al., 2016; Krauss et al., 2018). Further, there are currently no consistent degradation-specific emission factors available to estimate loss of carbon sequestration potential as there is for change in carbon stocks resulting from distinct mangrove deforestation causes (e.g., Sasmito et al., 2019).

Based on the current reference value of 2.81 MgC ha^{-1} yr^{-1} (**Table 4**), we estimated an annual loss of 10.31 $MgCO_2$ ha^{-1} yr^{-1} that would otherwise be buried in mangrove soils. Combined with loss of carbon sequestration potential in woody biomass, based on the current reference value of 3.18 MgC ha^{-1} yr^{-1} (**Table 3**), foregone carbon sequestration in Brazilian mangroves annually could total 22 $MgCO_2$ ha^{-1} yr^{-1}, in line with estimates reported for other mangroves worldwide (23-254 $MgCO_2$ ha^{-1} yr^{-1}as reviewed in Alongi, 2014).

TABLE 5 | Median values (95% Confidence Intervals) for potential CO_{2eq} emissions resulting from mangrove forest loss across Brazil.

State	Dominant driver of mangrove loss[a]	Potential emissions AGB ($MgCO_{2eq}$ ha^{-1})	Potential emissions SOC ($MgCO_{2eq}$ ha^{-1})	Potential emissions Ecosystem-level ($MgCO_{2eq}$ ha^{-1})
Alagoas (AL)	Erosion	169 (154–185)	1,040 (1,030–1,050)	1,210 (1,190–1,220)
Amapá (AP)	Erosion	376 (348–397)	767 (508–768)	1,030 (911–1,110)
Bahia (BA)	Clearing	130 (109–139)	214 (212–215)	341 (326–350)
Ceará (CE)	Clearing	97 (91–113)	195 (186–195)	288 (283–315)
Espírito Santo (ES)	Settlement	208 (180–223)	707 (620–736)	885 (832–921)
Maranhão (MA)	Erosion	291 (279–298)	655 (639–657)	937 (919–957)
Pará (PA)	Erosion	358 (349–363)	719 (633–767)	1,080 (1,010–1,110)
Paraíba (PB)	Clearing	97 (92–102)	207 (206–207)	307 (299–309)
Paraná (PR)	Extreme weather	53 (49–58)	138 (134–138)	193 (189–195)
Pernambuco (PE)	Agri/Aquiculture	143 (135–176)	536 (527–536)	677 (657–713)
Piauí (PI)	Erosion	251 (140–316)	877 (871–877)	1,120 (1,010–1,190)
Rio de Janeiro (RJ)	Erosion	145 (134–152)	1,070 (1,060–1,120)	1,250 (1,200–1,260)
Rio Grande do Norte (RN)	Agri/Aquiculture	148 (134–153)	519 (511–519)	667 (651–672)
Santa Catarina (SC)	Agri/Aquiculture	82 (64–95.)	545 (533–556)	620 (612–655)
São Paulo (SP)	Agri/Aquiculture	122 (111–128)	516 (513–519)	645 (625–649)
Sergipe (SE)	Clearing	120 (106–148)	220 (218–220)	340 (327–373)
Total		2,791	8,925	11,590

[a]After Goldberg et al. (2020). See "Materials and Methods" section for details about emission factors applied to estimate CO_{2eq} emissions for each of these categories. AGB, aboveground biomass; SOC, soil organic carbon.

CONCLUSION AND RECOMMENDATIONS

Here we deliver the first integrated assessment of mangrove carbon stocks, carbon sequestration rates and potential CO_{2eq} emissions for each Brazilian state. While more data are needed (e.g., particularly on carbon sequestration and emission factors) to better quantify national level statistics, this study provides compelling information to both aid the inclusion of mangroves in national (or state-level) carbon credit markets and establish Brazilian mangroves as hotspots within the context of global blue carbon policies. Our estimates suggest that Brazilian mangroves can potentially release substantial amounts of carbon following mangrove forest loss, with CO_{2eq} emissions nearing those estimated for other carbon-rich mangrove forests. In addition, loss of carbon sequestration potential in both woody biomass and soils following deforestation amplifies cumulative emissions annually, shortening the country's capacity to mitigate its fossil fuel emissions and meet intended NDC's.

In summary, we showed that Brazil is home of 9.3% of the world's mangroves, commensurably holding 8.5% of the global mangrove carbon stocks (biomass and soils combined). When compared to other Brazilian vegetated biomes, on a per-area basis mangroves store between 2.2 and 4.3 times more carbon in the top meter of soil. While for carbon stocks in biomass, Brazilian mangroves are second only to the Amazon forest, and store between 2.7 and 4.7 times more carbon than other vegetated biomes. Moreover, on a per-area basis organic carbon sequestration rates in Brazilian mangroves are 15–30% higher than recent global estimates. Importantly, integrated over the country's area, carbon sequestration in Brazilian mangroves soils account for 13.6% of the carbon buried in world's mangroves annually. Likewise, carbon sequestration in Brazilian mangroves woody biomass is also higher than global estimates, accounting for nearly 10% of carbon accumulation in mangrove woody biomass globally.

This study also highlights important research gaps and uncertainties in Brazilian mangroves carbon inventories. For example, the greatest carbon sink capacity in mangroves lies in the soils since this ecosystem compartment continuously fixes and preserves layers of millennia-old atmospheric carbon beneath the surface. However, we still know very little about the carbon sequestration potential of Brazilian mangroves soils, particularly the contribution of the Amazon Macrotidal Mangrove Coast (AMMC) to global carbon budgets. To date, we have found only one study reporting on soil carbon sequestration rates in this region (**Table 4**). Overall, several of Brazil's northern and northeastern states, where >80% of the country's mangroves are present, lack data; seven and nine states out of the 16 mangrove states in Brazil still lack data on soil organic carbon stocks and sequestration rates, respectively (**Supplementary Table 1**). It should also be noted that, while this study focused on carbon stocks and long-term carbon sequestration rates in biomass and soils, real time air-sea CO_2 fluxes, and DOC (dissolved organic carbon), DIC (dissolved inorganic carbon), and alkalinity (bicarbonate) export are important mechanisms of the carbon cycling in mangroves (e.g., Sippo et al., 2016; Carvalho et al., 2017; Cotovicz et al., 2019; Cabral et al., 2021) and should be taken into account to better constrain carbon budgets for Brazilian mangroves.

Mangrove AGB density has been consistently mapped across Brazilian mangroves, but disparities exist. For instance, no data was apparent for Alagoas' mangroves and only a few plots have been implemented in Amapá (2 plots), Piauí (2 plots), Rio Grande do Norte (2 plots), and Paraíba (6 plots) states (**Supplementary Table 1**). While for carbon sequestration in woody biomass, currently only two states (Rio de Janeiro and São Paulo) are represented (**Table 3**). The situation is far more critical for BGB density and productivity estimates. In this study we used a 0.5 BGB:AGB ratio to estimate BGB across Brazilian mangroves; however, to our knowledge, there are only two studies that have comprehensively (using trenching vs. coring techniques; see Adame et al., 2017 for a comprehensive discussion) assessed actual BGB distribution in Brazilian mangroves (Santos et al., 2017 in Rio de Janeiro and Virgulino-Júnior et al., 2020 in Pará). Moreover, BGB productivity and root necromass, which are important contributors to refractory carbon stored in mangroves soils (Kihara et al., 2021), remain unknown for Brazilian mangroves.

It is imperative that future research efforts and funding opportunities focus on addressing these data coverage issues. This is particularly pertinent for the data-poor northern states, where the AMMC is located, as carbon fluxes are more intense due to the synergistic contribution or riverine and tidal forcings that dictate coastal and ecological processes (e.g., deposition, erosion, mineralization, export). We recommend future carbon inventories in Brazilian mangroves to look beyond carbon stocks in biomass and soils and prioritize carbon fluxes via biomass (e.g., woody biomass growth) and soils (long-term carbon sequestration) as well as export of other carbon forms (e.g., DOC, DIC, alkalinity), which provide a direct comparison to greenhouse gases emission rates. Overall, this study consolidates the scientific basis demonstrating the significance of Brazilian mangroves to achieve NDC's both by enforcing environmental regulations to protect the country's existing mangroves and by promoting mangrove restoration where feasible to increase carbon crediting potential.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author.

AUTHOR CONTRIBUTIONS

AR conceived the study, collated and analyzed data, and wrote the draft. RT, TW, and PR analyzed data and wrote the draft. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/ffgc.2021.787533/full#supplementary-material

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Potential for Return on Investment in Rehabilitation-Oriented Blue Carbon Projects: Accounting Methodologies and Project Strategies

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Opportunities to boost climate change mitigation and adaptation (CCMA) and sustainable conservation financing may lie in enhancing blue carbon sequestration, particularly in developing nations where coastal ecosystems are extensive and international carbon markets offer comparatively attractive payments for environmental stewardship. While blue carbon is receiving increased global attention, few credit-generating projects are operational, due to low credit-buyer incentives with uncertainty in creditable emissions reductions and high project costs. Little empirical guidance exists for practitioners to quantify return-on-investment (ROI) and viability of potential projects, particularly for rehabilitation where multiple implementation options exist with diverse associated costs. We map and model drivers of mangrove natural regeneration (NR) using remote sensing (high-resolution satellite imagery segmentation and time-series modeling), and subsequent carbon sequestration using field- and literature-derived data, across abandoned aquaculture ponds in the Philippines. Using project-specific cost data, we then assess ROI for a hypothetical rehabilitation-focused mangrove blue carbon project at a 9.68 ha abandoned pond over a 10-year timeframe, under varied rehabilitation scenarios [NR vs. assisted natural regeneration (ANR) with planting], potential emissions reduction accreditation methodologies, carbon prices and discount rates. NR was faster in lower-lying ponds with lower tidal exposure (greater pond dike retention). Forecasted carbon sequestration was 3.7- to 5.2-fold and areal "greenbelt" regeneration 2.5- to 3.4-fold greater in our case study under ANR than NR. Variability in modeled sequestration rates drove high uncertainty and credit deductions in NR strategies. ROI with biomass-only accreditation was low and negative under NR and ANR, respectively. ROI was greater under ANR with inclusion of biomass and autochthonous soil carbon; however, neither strategy was highly profitable at current voluntary market carbon prices. ANR was the only scenario that fulfilled coastal protection greenbelt potential, with full mangrove cover within 10 years. Our findings highlight the benefits of ANR and soils inclusion in rehabilitation-oriented blue carbon projects, to maximize carbon sequestration and greenbelt enhancement (thus enhance pricing with potential bundled credits), and minimize forecasting uncertainty and credit-buyers' perceived risk. An ANR rehabilitation strategy in low-lying, sea-facing abandoned ponds with low biophysical intervention costs may represent large blue carbon CCMA opportunities in regions with high aquaculture abandonment.

Keywords: mangroves, carbon emissions reduction, rehabilitation, natural regeneration, blue carbon, remote sensing

INTRODUCTION

Coastal ecosystems, such as mangrove forests, are among the world's most productive ecosystems, maintaining high levels of biodiversity (Thompson and Rog, 2019) and delivering substantial ecosystem services to support local- to global-scale human well-being relative to their spatial coverage (Donato et al., 2011; McLeod et al., 2011; Curnick et al., 2019). Particularly pertinent in the current global climate emergency is their ability to support climate change mitigation and adaptation (CCMA) across the world's coasts, due to high relative sequestration and storage of "blue carbon," and protecting coastal communities and infrastructure from increasingly frequent storm conditions (Donato et al., 2011; Lee et al., 2014; Duncan et al., 2016; Hochard et al., 2019). Despite their importance, mangroves remain in global decline due to high coastal land-use demand and extractive dependency (Richards and Friess, 2016; Thomas et al., 2017; Bunting et al., 2018; Friess et al., 2019), facing substantial future challenges from abiotic climate change processes (Lovelock et al., 2015; Ward et al., 2016). In some regions, extensive historical mangrove loss to production land-uses such as aquaculture ponds (Richards and Friess, 2016; Kauffman et al., 2017; Bunting et al., 2018; Goldberg et al., 2020) has led to their rehabilitation being high on national coastal management and conservation agendas (Primavera and Esteban, 2008; Phan et al., 2015; Lee et al., 2019). To promote incentives for conservation intervention, increasing empirical scientific focus is now placed on identifying investible mangrove ecosystem service benefits, in particular for blue carbon projects (e.g., The Blue Natural Capital Financing Facility, 2021; Zeng et al., 2021). Such opportunities could unlock sustainable conservation financing, particularly in low- and middle-income nations where international carbon markets could offer comparatively attractive payments for environmental stewardship (Thompson et al., 2014). However, high perceived risk in blue carbon permanence, uncertainty in creditable emissions forecasting in the absence of blue carbon-specific quantification methodologies, large project costs and political risk have meant that mangroves' high CCMA potential has historically been largely unrealized in terms of operational blue carbon projects (Locatelli et al., 2014; Wylie et al., 2016; Herr et al., 2017).

In 2020, a major milestone was reached with the first blue carbon emissions reduction/sequestration quantification methodology approved under the Verified Carbon Standard that now enables inclusion of disproportionately large carbon sequestration in the soil compartment of blue carbon ecosystems (Verra, 2020a). This is likely to now herald a wealth of emerging blue carbon projects globally. To capitalize on this opportunity, a challenge to the design and implementation of blue carbon projects now lies in the ability of practitioners to assess potential return-on-investment (ROI), and hence their viability, prior to embarking on extensive and costly project registration and verification processes. While a major driver of potential blue carbon project costs is likely to be spatial scale, variation in initiation and on-going budget requirements will also depend on project implementation design. This is particularly true for rehabilitation-oriented mangrove management, where multiple intervention options exist with diverse associated costs and probability of long-term success (Bayraktarov et al., 2016; Lee et al., 2019; Wodehouse and Rayment, 2019; Su et al., 2021). Managers may adopt Ecological Mangrove Rehabilitation, with interventions to reinstate former hydrology in converted lands (e.g., abandoned aquaculture ponds, salt ponds, and agricultural land) to facilitate natural mangrove regeneration [hereafter "natural regeneration" (NR); Lewis and Brown, 2014], or more costly Assisted Natural Regeneration (hereafter "ANR"). ANR employs (a) (community-based) out-planting of nursery-reared or naturally available wildings to supplement NR in sites with more challenging or sub-optimal conditions, and (b) optional restoration of hydrology [e.g., breaching of aquaculture pond banks ("dikes") where required] (Primavera et al., 2012b; Mangrove Action Project, 2021). Either strategy may be optimal for a given site, with faster and denser mangrove regeneration enhancing credit generation, associated ecosystem services (e.g., coastal protection, fisheries enhancement), and therefore pricing under co-benefit accreditation schemes (Plan Vivo, 2013). However, quantitative evaluation of which rehabilitation strategy may be financially optimal in a given context remains challenging. Research to date has quantified investible blue carbon opportunities and ROI utilizing broad mean project costs across large spatial scales and contexts (e.g., Jakovac et al., 2020; Taillardat et al., 2020; Zeng et al., 2021). However, the relative ecological outcomes of mangrove NR and ANR strategies are rarely considered in empirical studies, and these authors know of no existing study to date that explores variation in blue carbon credit generation potential and ROI under differing rehabilitation intervention strategies. This empirical data gap presents a particular challenge for managers designing potential small-scale (and community-based) blue carbon projects with minimal implementation budgets. There is thus a significant gap between the theory and practice of mangrove blue carbon projects that may represent a barrier to the development of operational projects to date despite the high interest.

The guiding principles underpinning successful implementation of NR-oriented rehabilitation are tidal connectivity to a viable source of mangrove propagules, favorable intertidal position [i.e., elevation above mean sea level (a.m.s.l.)] and low tidal energy exposure for rapid propagule establishment and regeneration in former mangrove areas (Lewis and Brown, 2014). By contrast, ANR can bolster NR rates in favorable sites (see also Huxham et al., 2010) and/or supplant NR where one or more of these conditions is sub-optimal, thus enhancing mangrove regeneration rates and strengthening coastal protection ("greenbelts") in the face of rapidly advancing climate change (Primavera et al., 2012b; Mangrove Action Project, 2021). However, ANR can incur substantial costs (Primavera et al., 2012b; Bayraktarov et al., 2016), which must be weighed against the relative potential carbon credit returns vs. NR alone (i.e., ROI) to assess whether the approach is justified. Drivers of mangrove NR have been extensively established in experimental and site-specific field studies (e.g., Balke et al., 2011; Kamali and Hashim, 2011); however, their empirical quantification across larger scales (e.g., between sites) is absent, hindering out ability to predict (rates of) site-specific carbon sequestration potential from NR relative to ANR. Furthermore, ANR may buffer natural inter-annual variability in propagule availability and/or stochastic perturbations, thereby reducing uncertainty in emissions reduction forecasting and associated credit reductions (see Verra, 2019), but this remains empirically unquantified across rehabilitation projects. These data gaps may be a reason that all currently registered blue carbon projects do not fully account for NR in their *ex ante* project emissions reduction forecasts [ANR strategies only: Blue Ventures, 2019; Mikoko Pamoja, 2020; rewetting/soil stabilization from reinstated hydrology only: Conservation International, 2021]. Remote sensing has long been applied in largely inaccessible mangroves to successfully track changes in their distribution and functioning (Wang et al., 2019), and high resolution imagery has recently been employed to track fine-scale (tree-specific) landward mangrove expansion (Whitt et al., 2020). Similar high resolution monitoring of NR and ANR rates combined with spatial analysis across landscape-scales may enable quantification of NR rate drivers and relative rehabilitation and emissions reduction potential under similar conditions. However, to date, these approaches have not been employed to enhance our predictive capacity in predicting carbon sequestration in areas under different management regimes and in evaluating potential blue carbon project scenario options.

To close these identified data gaps and investigate the relative efficacy of ANR over NR-oriented mangrove blue carbon projects, here we model potential carbon sequestration and credit generation potential in a hypothetical project using a case study abandoned aquaculture pond in West Visayas, Philippines. The Philippines has experienced substantial mangrove loss: approximately 50% of the former 500,000 ha (Spalding et al., 2010) disappeared over the last century, due primarily to "extensive" (large area but shallow depth) brackish-water pond aquaculture development in former mangrove areas (Primavera, 2005). Some of the highest pond densities occur in the West Visayas region (Primavera and Esteban, 2008; Department of Agriculture of the Philippines – Bureau of Fisheries and Aquatic Resources, 2020). Development is largely unregulated, and despite laws mandating 50–100 m greenbelts (Primavera et al., 2012a), ponds are often built to the shoreline. Abandonment is high (Samson and Rollon, 2011; Primavera et al., 2012a), due primarily to bank ("dike") breaches in sea-facing ponds over low productivity (Primavera et al., 2014). Legal mandates in the Philippines dictate that government-leased ponds that are abandoned must be reverted to public lands and rehabilitated to mangrove forest by the relevant government department. However, a myriad of confounding factors, centered predominantly on low political will, means cancelation and reversion rarely occurs: large areas of ponds built in former mangrove lie fallow and the few canceled leases are often absorbed and re-tenured or operated illegally, thus the long-term persistence of any naturally regenerated (NR) mangroves within abandoned ponds is not secured (Primavera et al., 2014). If better regulated and enforced, abandoned pond tenure reversion could afford a major rehabilitated-oriented emissions reduction opportunity in the Philippines, with minimal physical intervention required to generate blue carbon additionality (natural dike breaching; Primavera et al., 2014). Herein, we explore the potential for this abandoned pond blue carbon opportunity under NR and assisted NR (ANR; e.g., planting) strategies. First, we map mangrove NR (n = 8) and ANR (n = 1) at annual time-steps at identified abandoned aquaculture ponds, using a novel image classification approach with open access high resolution imagery. Second, we use open access spatial data to model the main drivers of these time-series NR rates of mangrove areal coverage across abandoned ponds. Third, we employ model-derived and observed NR and ANR rates (and uncertainty) to predict mangrove regeneration across a hypothetical 10-year time period at our case study abandoned pond, and apply field- and literature-derived rehabilitated mangrove carbon sequestration data to forecast potential rehabilitated carbon stocks at the site under these scenarios. Finally, we apply approved emissions reduction quantification protocols (Verra, 2020b) under potential accreditation methodologies, alongside ANR project cost data, varied potential voluntary market carbon prices and discount rates, to explore variability in relative ROI across NR and ANR scenarios at our case study site. Here we hypothesize slower (and more variable) areal regeneration and carbon sequestration under NR, and greater ROI with ANR over NR after deduction of biophysical project costs.

MATERIALS AND METHODS

Site Selection

Via inspection of high-resolution Google Earth (GE) Red-Green-Blue (RGB) imagery for 2018–2019 (Google Earth Pro, 2019), we selected NR abandoned aquaculture ponds (those with mangroves present) in West Visayas (**Figure 1**) as study sites under the following criteria: (1) they were located in the

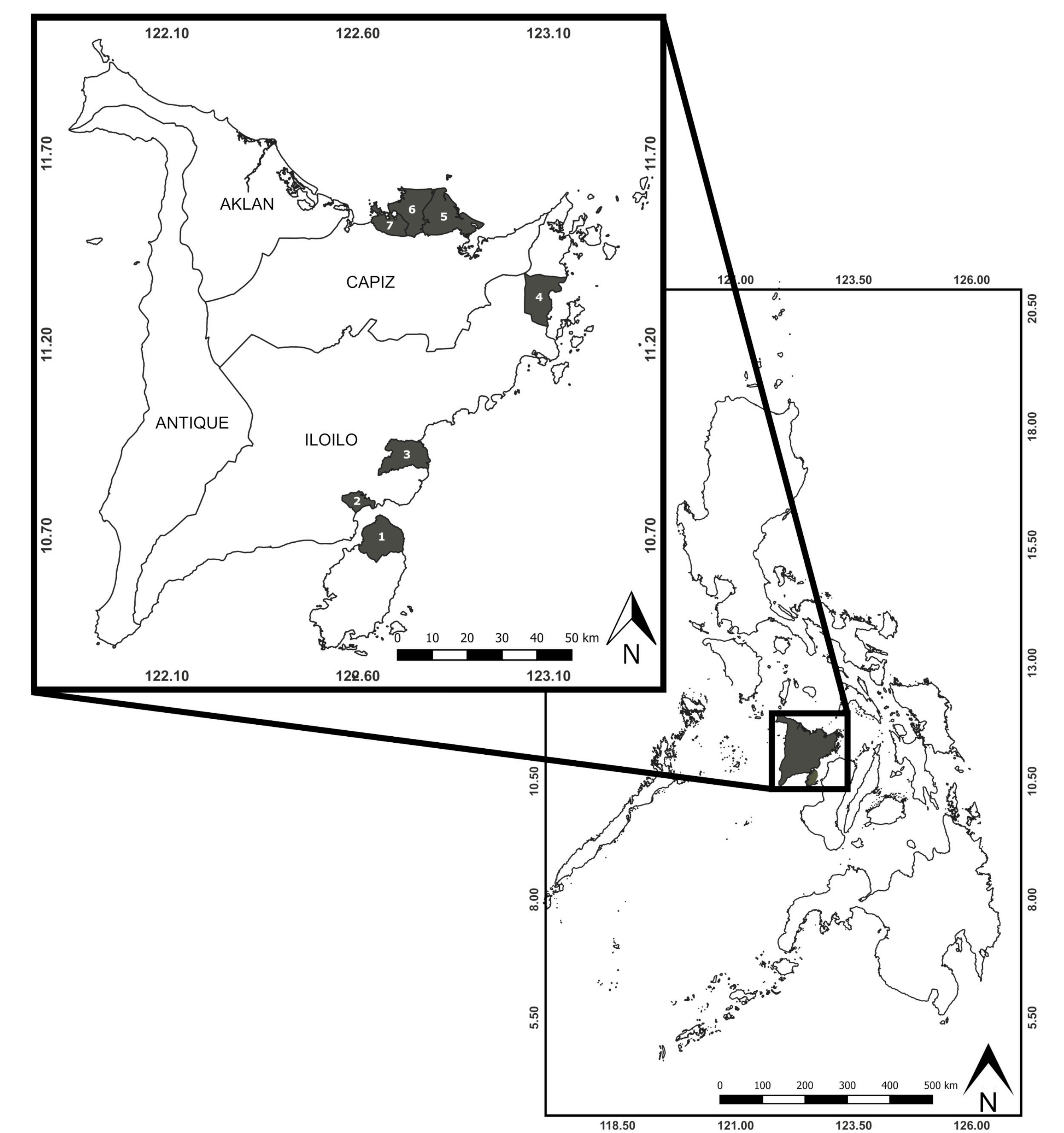

FIGURE 1 | Locations of municipalities with abandoned aquaculture ponds examined within this study, within Iloilo and Capiz Provinces, West Visayas, Philippines. (1) Buenavista, Guimaras, Iloilo [n = 1 naturally-regenerating (NR) abandoned aquaculture pond]; (2) Leganes, Iloilo [n = 1 NR and n = 1 assisted naturally-regenerating (ANR; planted) abandoned pond]; (3) Barotac Nuevo, Iloilo (n = 1 NR abandoned pond); (4) San Dionisio, Iloilo (n = 1 NR abandoned pond); (5) Panay, Capiz (n = 1 NR abandoned pond); (6) Roxas City, Capiz (n = 1 NR abandoned pond), and; (7) Ivisan, Capiz (n = 1 NR abandoned pond).

coastal zone of Panay or Guimaras islands; (2) they were sea- or estuary-facing (breached pond dikes immediately adjacent to these water bodies), i.e., they had direct hydrological access for moderate-distance propagule dispersal from adjacent established mangrove stands; (3) there was high-resolution GE RGB imagery coverage from the point of sea- or estuary-facing dike breaching (e.g., beginning of the process of NR), and; (4) at least three high-resolution RGB imagery time-points were available post-dike breaching. Under these criteria, we identified seven abandoned ponds with high-resolution RGB imagery to quantify mangrove NR over time, covering Buenavista, Leganes, Barotac Nuevo, and San Dionisio municipalities in Iloilo Province, and Panay, Roxas City, and Ivisan municipalities in Capiz Province (**Figure 1**).

In addition, we selected one abandoned pond site, Leganes Katunggan ecopark, Iloilo, known to have experienced early NR (2005–2009) followed by ANR. Here, sea-facing pond dikes were breached by wave action between 2005 and 2008, enabling tidal and mangrove dispersal re-connection, and in 2009 tenure status of the pond was secured by the Leganes local government unit (LGU; Primavera et al., 2012b). ANR consisted of initial out-planting of ∼20,000 seedlings (mainly *Avicennia marina*, with some *Sonneratia alba* and *Rhizophora* sp.) in 3.5 ha of the mid- to upper-intertidal pond zone over 2009–2011 under the Zoological Society of London (ZSL) Philippines' Community-based Mangrove Rehabilitation Project, followed by further planting of ∼63,000 seedlings over the remaining 6.18 ha over 2012–2015 under Leganes LGU governance. Small-scale voluntary planting activities continue in the erosion-prone sea-facing area of the pond post-2015 (Loma and Batislaon, 2021, *pers. comm.* 2021). In all cases, rehabilitation labor and expenses for wilding collection and out-planting were provided through voluntary activity from student groups, local fisherfolk, government employees and civil society organization groups, and seedling nursery and plantation maintenance conducted in collaboration between local fisherfolk and Leganes LGU. For further rehabilitation and site details see Primavera et al. (2012b) and Duncan et al. (2016).

High-Resolution Imagery Acquisition, Sub-Setting, and Pre-processing

For each identified abandoned pond, we exported all available historical RGB GE imagery at the highest possible spatial resolution from the point of pond dike breaching onward (to 1st August 2019). We reduced initial high-resolution RGB imagery datasets for each abandoned pond to retain a single image for each available year since dike breaching, with, where possible, the selected annual (day-time) image: (1) having no cloud cover present [three sites had one image in the time-series with partial cloud cover: Basiao, Ivisan (2008: 25.00% of images), Matnog, Ivisan (2013: 14.29% of images) and Talon, Roxas City (2013: 11.11% of images)]; (2) being obtained at low-tide [22 of 55 images were acquired at high tide; mean 39.97% (0-75.00%) per site], and; (3) being acquired within a similar time of year as all other imagery within a site's time-series (see **Supplementary Table 1**). In addition to the original RGB bands, we derived additional indices using the "jpeg" (Urbanek, 2019) and "raster" (Hijmans et al., 2019) packages in R v.3.6.1 (R Development Core Team, 2019): The Green-Red Vegetation Index (*GRVI*), an index describing vegetation "greenness" from RGB imagery (Motohka et al., 2010) in the absence of near-infrared bands in the initial imagery to quantify, e.g., the Normalized Difference Vegetation Index (Pettorelli, 2013), and a "high reflectance" index (*Red* + *Green* + *Blue*), to aid in identifying features such as man-made structures and bare ground. Outputted stacked rasters (all raw bands and indices: n = 5) for each time-point were then georeferenced in QGIS v.3.8.3 (QGIS Development Team, 2019), to 1 m^2 resolution, against site-specific ground control points. Ground control points were created in GE, using the most recent available high-resolution RGB imagery, using easily identifiable features (e.g., corners of pond dikes, buildings, and centers of trees) that were static through entire time-series [mean 9.44 (5-18) points per site].

Imagery Segmentation and Classification

Time-point-specific georeferenced stacked rasters for each abandoned pond were then split into individual band and index rasters, and Simple Linear Iterative Clustering (SLIC) superpixels segmentation (Achanta et al., 2012) was applied across all site-specific rasters (n = 5) for each time-point in SAGA GIS v.7.3.0 (Conrad et al., 2015). The starting superpixel size was set to four pixels (4 m^2), with a minimum output segment size of one pixel (1 m^2), and segmentation conducted over a maximum of 100 iterations per time-point. Owing to low spectral resolution in input imagery (e.g., RGB and two derived indices only), classification algorithms were not applied to outputted segmented imagery; instead, segments representing mangroves were manually extracted via inspection against original high-resolution RGB GE imagery (≥25% of a segment with mangrove present) in QGIS. While we acknowledge that this approach possibly overestimated mangrove cover, the same rule was applied to all imagery, thus not impacting estimated rates of mangrove regeneration within- and between-abandoned ponds. Time-point-specific extracted mangrove segments were then dissolved, and areal estimates of mangrove cover calculated.

Variable Creation: Potential Drivers of Natural Regeneration Rates

Two variables were created to proxy local-scale propagule availability, using Philippines' extant mangrove cover in 2010 (Long et al., 2014): "adjacent mangrove extent" and "distance to source population" (**Table 1**). To index "adjacent mangrove extent," 2010 mangrove cover (hectares) was clipped to a buffer of 2.5 km from the boundaries of each abandoned pond and its area extracted (R package "rgeos": Bivand et al., 2019). To index "distance to source population," 2010 mangrove cover was analyzed for contiguous patches ("clump" function: Hijmans et al., 2019), polygonized in QGIS, and the minimum distance between each abandoned pond and its nearest mangrove patch ≥10 ha in area extracted (Bivand et al., 2019). Two variables were created to proxy exposure to wave energy (propagule and sediment retention) within abandoned ponds, using digitized sea- or estuary-facing pond dikes from the most recent image in the respective time-series of high-resolution RGB GE imagery: "proportional remaining pond dikes" and "relative dike protection" (see **Table 1** for calculation methods). Finally, two variables were created to proxy (variation in) ground elevation, using 30 m resolution Shuttle Radar Terrain Model (SRTM) Digital Elevation Model (DEM) data (year 2000:

TABLE 1 | Variables employed in statistical analysis of potential drivers of natural regeneration (NR) rates in abandoned aquaculture ponds.

Proxy process	Variable	Calculation	Source data	Hypothesized influence on NR
Local-scale propagule availability	Adjacent mangrove extent (ha)	Extant mangrove area (year 2010) within a 2.5 km buffer of abandoned pond extent	Long et al., 2014	Positive
	Distance to source population (km)	Distance (km) to a contiguous extant mangrove patch (year 2010) of $\geq$10 ha	Long et al., 2014	
Propagule (and sediment) retention	Proportional remaining pond dikes	$\frac{\text{remaining sea or estuary dikes }(km)}{\text{total initial sea or estuary dikes }(km)}$	Google Earth Pro, 2019	Positive
	Relative dike protection	$\frac{\text{total initial pond area }(m^2)}{\text{remaining sea or estuary dikes }(m)}$	Google Earth Pro, 2019	
Mean elevation	Mean pond elevation (a.m.s.l.)	Mean SRTM DEM pixels' elevation a.m.s.l. (m) within abandoned pond extent	United States Geological Survey, 2014	Negative
Variation in elevation	Variation in pond elevation (a.m.s.l.)	Coefficient of variation (CV) in SRTM DEM pixels' elevation a.m.s.l. (m) within abandoned pond extent	United States Geological Survey, 2014	Negative

"Remaining sea or estuary dikes" = remaining length of sea- or estuary-facing abandoned pond dikes mapped from the latest year in the available time-series high-resolution RGB GE imagery; "total initial sea or estuary dikes" = total length of sea- or estuary-facing abandoned pond dikes mapped in the first year in the available time-series high-resolution RGB GE imagery; "total initial pond area" = total abandoned pond area mapped in the first year in the available time-series high-resolution RGB GE imagery; m.s.l., mean sea level.

United States Geological Survey, 2014): "mean pond elevation" (m: mean elevation across all SRTM DEM pixels within each abandoned pond extent) and "variation in pond elevation" [coefficient of variation (CV) in elevation (m) across all SRTM DEM pixels within each abandoned pond extent] (**Table 1**).

Data Analysis: Drivers of Natural Regeneration Rates

To establish rates of mangrove regeneration (areal increase) over time, we first constructed abandoned pond-specific time-series linear regression models with mangrove area (ha) as a response variable and year as the explanatory variable. Linear models predicting rate of areal increase were conducted only across years until which abandoned pond-specific areal increase "leveled-off." "Leveling-off" of rates of areal increase were established via the following approach for each abandoned pond: (1) the "peak maxima rate" (*PMR*) and "pre-peak minima rate" (*PPMR*: minimum observed rate of areal increase prior to the *PMR*) (ha year^{-1}) across the time-series were identified; (2) if the *PMR* was located at the end of the time-series or fewer than two rate datapoints were observed post-*PMR*, no "leveling-off" was identified; (3) if there was no *PPMR* in the time-series, "leveling-off" was only established when a post- *PMR* rate reached below the 25% quantile of the observed distribution of rates across the time-series (*Q*25); (4) if a post-*PMR* continuing decline in rate was observed, "leveling-off" was established immediately after the *PMR* if all post-*PMR* rates were below the *PPMR*; (4) if additional rate increases above the PPMR were observed post-*PMR*, "leveling-off" was established at the point at which the rate dropped below *Q*25 post- the final peak, and; (5) if additional rate increases below the *PPMR* were observed post-*PMR*, "leveling-off" was established at the first point at which the rate dropped below *Q*25 post-*PMR*. We acknowledge that conducting pond-specific time-series analyses only to the point of "leveling-off" reduced our linear regression sample sizes; however, ponds (and high-resolution imagery availability) were not consistent in time since dike breaching (or ANR) and analysis of full time-series datasets would have artificially deflated quantified NR rates in some ponds (i.e., those with longer time-series data available; see **Supplementary Table 1**) where mangrove cover NR rates decrease over time.

Two proxy response variables were then employed to explore drivers of rates of NR in abandoned ponds: "rate of areal increase" (mean slope estimates from the initial regressions for each pond: ha year^{-1}) and "time-to-leveling" (number of years post dike-breaching at which areal increase rates were observed to "level-off"). A list of candidate linear models was created for each response variable: (1) no interactions were anticipated between explanatory variables, and so all candidate models included additive terms; (2) variables proxying the same process (i.e., "adjacent mangrove extent" (ha) and "distance to source population" (km), and "proportional remaining pond dikes" and "relative dike protection") were not considered within the same models, and; (3) to avoid overparameterization of model fits, only three- and single-variable models were constructed for "rate of areal increase" (ha year^{-1}; n = 8 observations; 27 candidate models) and "time-to-leveling" (years; n = 4 observations; six candidate models) response variables, respectively. Models were ranked via AICc values (Akaike, 1974; Hurvich and Tsai, 1989), and best-fitting model(s) for each response variable were selected based on a threshold of delta AICc < 2 (Burnham and Anderson, 2002). Best-fitting model(s) for each response variable were then employed to site data for the Leganes Katunggan pond, to predict long-term rates of NR at the abandoned pond without planting activities.

Data Analysis: Carbon Sequestration and Credit Generation Potential Under Rehabilitation Strategies

We calculated carbon sequestration from mangrove NR and ANR in the Leganes Katunggan abandoned pond over a 10 year period, representing 2005–2015, under three scenarios: (1) best-fitting model-predicted "rate of areal increase" (ha year^{-1}) (run using mean model estimates minus and plus 1 s.e.),

(2) observed "rate of areal increase" under NR (2005–2009), and (3) observed "rate of areal increase" under ANR (planting: 2009–2013) at the site (run using ANR Leganes Katunggan-specific mean time-series model estimate plus and minus 1 s.e.). For each scenario, additional mangrove cover was applied at annual increments under the model-predicted or observed rates above. Annual carbon sequestration rates in soil and biomass compartments were applied to regenerated areas at each time-step. The low-intertidal, sea-facing, fringing nature of the studied abandoned ponds lends their soils to trapping relatively high levels of allochthonous (non-mangrove) organic carbon (Kusumaningtyas et al., 2019; Sasmito et al., 2020). Published soil carbon sequestration rates observed in NR fringing abandoned aquaculture ponds in Southeast Asia (0-10 years) were used in NR scenarios (1-2), assuming 1 cm $year^{-1}$ accretion rates (Sidik et al., 2019) [2.29 $\pm$ 0.96 (1 s.d.) Mg C ha^{-1} $year^{-1}$: Duncan et al., 2016 (Philippines); Salmo and Gianan, 2019 (Philippines); Sidik et al., 2019 (Indonesia)]. In NR scenarios, we applied published estimates of mangrove above- and below-ground biomass carbon sequestration (0-22 years) in NR abandoned ponds [1.18 $\pm$ 1.07 (1 s.d.) Mg C ha^{-1} $year^{-1}$: Duncan et al., 2016 (Philippines); Salmo and Gianan, 2019 (Philippines); Elwin et al., 2019 (Thailand)]. In the ANR scenario (3), we used Leganes Katunggan site-specific soil carbon sequestration rates, also assuming 1 cm $year^{-1}$ accretion rates, from 0 to 10 years post planting [3.27 $\pm$ 1.09 (1 s.d.) Mg C ha^{-1} $year^{-1}$: Duncan *unpublished data*], and above- and below-ground biomass carbon sequestration rates from 0 to 5 years post planting {1.56 $\pm$ 0.47 (1 s.d.) Mg C ha^{-1} $year^{-1}$: Duncan et al., 2016) and 5-10 years post planting [0.73 $\pm$ 0.83 (1 s.d.) Mg C ha^{-1} $year^{-1}$: Duncan *unpublished data*]. For each scenario (and under each areal cover increase rate (see above)}, we generated a range of predicted carbon stock estimates (site Mg C) over 2005-2015 from a normal distribution of the mean and s.d. of sequestration rates above (n = 30 per compartment). Under all scenarios, an initial year of carbon stock from existing mangrove vegetation [e.g., 2004-2005 mapped mangrove cover: 0.38 ha (2005 imagery)] at the above NR carbon sequestration rates was first added.

We converted site-level standing carbon stock projections to Mg CO_2e via application of a 3.67 conversion factor (Kauffman and Donato, 2012). We then calculated transactional carbon sequestration potential under three potential emissions reduction accreditation methodologies: *B*, only above- and below-ground biomass CO_2e emissions reduction creditable; *BAS*, biomass CO_2e and additional autochthonous soil organic carbon CO_2e emissions reduction creditable, and *TOT*, biomass CO_2e and all additional soil organic carbon CO_2e emissions reduction creditable. We calculated autochthonous soil organic carbon contributions for the *BAS* methodology via calculation of allochthonous organic carbon deductions using Needleman et al. (2018) from mean study sample soil organic carbon content (SOC; %). For NR scenarios (1, 2), mean literature-derived sample SOC (3.60%: Duncan et al., 2016; Sidik et al., 2019; Salmo and Gianan, 2019) was used to estimate autochthonous soil carbon contributions of 53.28% for the 10-year time-series. For the ANR scenario (3), we assumed that the literature-derived NR autochthonous carbon contribution estimate applied for years 0–5, and then applied field-derived mean sample SOC for the site at 10 years of age (5.94%: Duncan et al., *unpublished data*) and autochthonous soil carbon contributions of 74.14% for years 5–10, due to substantially increased mangrove biomass across this latter half of the time-series. We therefore used a mean of 63.71% autochthonous soil carbon contributions for the ANR scenario (3).

For each scenario and potential accreditation methodology, uncertainty in projected total (all relevant compartments) emissions reduction potential was calculated as:

$$uncertainty\ (\%) \ = 100 \times \left(\frac{95\%\ \mathrm{CI}\ PER}{mean\ PER} \right)$$

where PER = projected total project emissions reductions (CO_2e). In accordance with approved blue carbon methodologies (VM0007 revision for tidal wetlands: Verra, 2020b), deductions of creditable emissions were then made in any scenario for which the allowable precision level (15%) was exceeded, as:

$$Adjusted\_PER \ = PER \times \left(1 - \left[\frac{uncertainty - 15}{100} \right] \right)$$

In the absence of >2 annual data points for mangrove cover in observed NR (2005–2009), uncertainty in scenario 2 was assumed equal to that observed in projected sequestration under scenario 1 (model-predicted NR). We deducted non-permanence credit withholding buffers (Verra, 2019, 2020b) of 30% from net carbon sequestration in ANR scenario 3 to reflect moderate natural risks at the site (typhoons), and increased this buffer to 35% to reflect potential increased erosion/scouring in sea-facing ponds without out-planted saplings (Huxham et al., 2010; Balke et al., 2011; Primavera et al., 2012b).

For each scenario and potential accreditation methodology, we calculated the number of carbon credits generated (units of 1 Mg CO_2e per credit) at annual increments and carried any carbon sequestration not having generated credits (<1 Mg CO_2e) over to the following year. We calculated transactional carbon credit potential (USD) via application of three different carbon prices and three potential discount rates at annual increments. For each simulated year, i, discounting was applied to potential ROI from potential annual credit sales ($dROI_{p,d}$) as:

$$dROI_{p,d,i} \ = Credits_i \times Price_p \times \frac{1}{(1 \times \ [1 + \ Discount_d])^i}$$

where, *Credits* = scenario- and potential accreditation methodology-specific mean number of carbon credits generated, *Price* = carbon price applied [p = USD \$2.51 (mean 2020 voluntary market price); USD \$9.70 (mean 2020 Afforestation/Reforestation project voluntary market price: Forest Trends' Ecosystem Marketplace, 2021), or; USD \$25.00 Mg CO_2e^{-1} (hypothetical inflated price for associated human well-being and biodiversity benefits and with demand assumed to rise to 500 Mt CO_2e $year^{-1}$: Turner et al., 2021)], and *Discount* = discount rate applied [d = 1.50% (Stern, 2007; Weitzman, 2007); 3.50% (HM Treasury, 2020), or; 4.25% (Nordhaus, 2017)]. We then summed simulated ROI for all years under each combination of rehabilitation scenario, potential accreditation methodology, carbon price and discount rate. In ANR scenario 3, we also removed the realized biophysical costs of rehabilitation incurred for wilding collection, nursery operation and out-planting for ~83,000 seedlings over 9.68 ha and 6 years at Leganes Katunggan pond (USD 3,835.68: **Supplementary Table 2**; Primavera et al., 2012b). This rehabilitation cost is substantially lower than the literature-derived median cost from projects in developing nations applied over the same area (USD 10,831.84: Taillardat et al., 2020), owing to strong community engagement, voluntary labor and the implementation of science-based ANR protocols (Primavera et al., 2012b) in the Leganes Katunggan rehabilitation efforts. In potential accreditation methodologies *BAS* and *TOT*, costs of soil carbon baseline quantification and 0–15 cm depth monitoring at 3 year intervals are also removed under all scenarios (USD 1,170.00; **Supplementary Table 2**). Under the assumption that realized costs of registration and verification are similar across hypothetical NR and ANR rehabilitation-oriented blue carbon projects, we do not include these costs. However, we stress that these costs are substantial (~USD 40,000 for project registration and verification over our hypothetical 9.68 ha case study over 10 years: Verra, 2020c).

RESULTS

Our analysis of identified abandoned ponds in West Visayas found within-pond natural mangrove regeneration (following dike breaching) from as early as 2005 (Leganes Katunggan) to 2013 (Panay, Capiz) (see **Table 2**, **Figure 2**, and **Supplementary Figures 1A–F**). Abandoned pond characteristics varied widely, with total pond area ranging from 1.58 (Basiao, Ivisan, Capiz) to 17.90 ha (Buenavista, Guimaras, Iloilo), adjacent mangrove extent (within a 2.5 km buffer) from 0.56 (Balagon and Napnud, Leganes, Iloilo) to 108.67 ha (Barotac Nuevo, Iloilo), distance to source population (nearest 10 ha patch) from 0.02 (San Dionisio, Iloilo) to 11.64 km (Roxas City, Capiz), proportional dikes remaining from 0.36 (Leganes Katunggan) to 0.95 (Buenavista, Guimaras), relative dike protection from 89.98 (Basiao, Ivisan, Capiz) to 526.35 m^2 m^{-1} (Leganes Katunggan), SRTM DEM-derived mean pond elevation from 0.42 (Roxas City, Capiz) to 3.63 m (San Dionisio), and variation (CV) in pond elevation from 17.34% (San Dionisio) to 341.39% (Balagon and Napnud, Leganes) (see **Table 2**).

"Rate of areal increase" under NR was highly variable across abandoned pond time-series, ranging from 0.077 ± 0.007 (1 s.e.) ha $year^{-1}$ at Basiao, Ivisan to 1.434 ± 0.146 (1 s.e.) ha $year^{-1}$ at Balagon and Napnud, Leganes. Among the highest "rate of areal increase" was observed during assisted natural regeneration [ANR (planting): 2009?2013] at the Leganes Katunggan pond [1.327 ± 0.157 (1 s.e.) ha $year^{-1}$]; however, this rate was exceeded under NR at three sites [Balagon and Napnud, Leganes; Barotac Nuevo: 1.339 ± 0.215 (1 s.e.) ha $year^{-1}$; Buenavista, Guimaras: 1.337 ± 0.198 (1 s.e.) ha $year^{-1}$] (**Table 2**, **Figures 2**, **3**, and **Supplementary Figures 1A–F**). "Leveling" of rates of areal increase was not reached in three non-case study NR abandoned ponds, possibly owing to recent (2012) and slow regeneration at San Dionisio, slow early regeneration at Roxas City, and very slow regeneration at Basiao, Ivisan (**Table 2**, **Figures 2**, **3**, and **Supplementary Figure 1**). "Time-to-leveling" for the remaining ponds was lowest (4 years) at the small (6.40 ha) and moderately quickly regenerating Panay abandoned pond and under ANR at the Leganes Katunggan pond (**Table 2** and **Figure 3**).

Drivers of Natural Regeneration Rates

In order to enhance predictive power, we selected our "best-fitting" model explaining variation in "rate of areal increase" (ha $year^{-1}$) across NR abandoned ponds as the more complex of two models with delta AICc < 2 (**Table 3**). This model included the negative effect of mean pond elevation (m a.m.s.l.) [intercept = 0.47 ± 0.35 (1 s.e.), β1 = –0.30 ± 0.06 (1 s.e.), *t1* = –5.18] and the positive effect of proportional remaining pond dikes [β2 = 1.12 ± 0.37 (1 s.e.), *t2* = 3.00] (5 d.f., p = 0.002, multiple R^2 = 0.91: **Table 3** and **Figure 4**). Mean model-predicted "rate of areal increase" under NR at the Leganes Katunggan abandoned pond (0.247 ha $year^{-1}$) was lower than that observed at the site under early NR (0.352 ha $year^{-1}$: 2005–2009), and substantially lower than that observed at the site under ANR (1.327 ha $year^{-1}$; 2009–2013) (**Figures 2**, **3** and **Table 2**). We found no significant effect of any explanatory variable in explaining variation in, and therefore no "best-fitting" model for, "time-to-leveling" (years) across NR abandoned ponds.

Carbon Sequestration and Transactional Credit Generation Potential Under Rehabilitation Strategies

Owing to variation in model-predicted and observed rates of NR and ANR (**Table 2** and **Figure 4**), simulated rehabilitated mangrove area at the Leganes Katunggan abandoned pond across the 2005–2015 hypothetical simulation period varied widely in the three rehabilitation scenarios. Predicted mangrove cover varied from 2.84 ha under the best-fitting model-predicted NR (*1*) to full pond cover (9.68 ha) under the ANR (*3*) scenarios at Leganes Katunggan (**Table 4**). Accordingly, predicted soil and biomass organic carbon sequestration across the three scenarios was similarly variable, for example with mean soil and biomass gain ranging from 40.65 to 20.98 Mg under the best-fitting model-predicted NR scenario 1 to 227.25 and 96.11 Mg under the observed ANR scenario 3, respectively (**Table 4**). Wide variation (±1 s.e.) in best-fitting model-predicted rates of areal increase at the site resulted in wide variation in predicted mangrove cover (range: 0.28-5.41 ha) and soil and biomass organic carbon sequestration (range: 4.23–118.36 and 0.00–119.18 Mg, respectively) under scenario 1. Total predicted soil carbon sequestration was in all cases substantially greater than total predicted biomass carbon sequestration (**Table 4**). Uncertainty in total potential project emissions exceeded the allowable uncertainty threshold (15%) under NR (scenarios 1, 2) for all potential accreditation methodologies: *B*, 26.28% (11.28% deduction applied); *BAS*, 21.38% (6.38% deduction applied); *TOT*, 20.01% (5.01% deduction applied). Uncertainty in total project emissions did not exceed the 15% allowable uncertainty threshold under ANR for any methodology.

TABLE 2 | Summary statistics for "rate of areal increase" and "time-to-leveling" and proxies of potential drivers of regeneration speed for all abandoned aquaculture ponds considered in this study.

Type	Site	Time span	Rate of areal increase (ha year^{-1})	Time-to-leveling (years)	Total initial pond area (ha)	Adjacent mangrove extent (ha)	Distance to source population (km)	Proportional remaining pond dikes	Relative dike protection (m^2 m^{-1})	Mean pond elevation (m)	Variation in pond elevation (CV; %)
Natural regeneration (NR)	Buenavista, Guimaras, Iloilo	2009–2017	1.337	8	17.90	2.37	5.41	0.95	153.55	0.58	224.87
	Balagon and Napnud, Leganes, Iloilo	2012–2017	1.434	5	10.60	0.56	2.55	0.94	341.39	1.03	341.39
	Barotac Nuevo, Iloilo	2012–2017	1.339	5	7.55	108.67	1.19	0.91	122.09	0.88	197.25
	San Dionisio, Iloilo	2012–2016	0.519	–	2.53	28.69	0.02	0.89	119.53	3.63	17.34
	Panay, Capiz	2013–2017	0.736	4	6.40	90.37	0.35	0.75	274.66	1.42	162.18
	Roxas City, Capiz	2009–2019	1.182	–	15.89	38.07	11.64	0.90	164.14	0.42	207.03
	Basiao, Ivisan, Capiz	2008–2016	0.077	–	1.58	10.39	5.94	0.76	89.98	3.59	106.60
	Leganes Katunggan, Iloilo (*pre-planting*)	2005–2009	0.352*	–	9.68	18.19	1.77	0.36	526.35	2.13	37.53
Assisted natural regeneration (ANR; planting)	Leganes Katunggan, Iloilo (*with planting*)**	2009-2013	1.327	4	9.68	18.19	1.77	0.36	526.35	2.13	37.53

N.B. time-series data for the Leganes Katunggan, Iloilo site under active rehabilitation [ANR (planting)] were not considered in regression analyses.
**"Rate of areal increase" (ha year^{-1}) calculated from a real change across two time-points (2005 and 2009 high-resolution imagery) and converted to an annual rate of increase.*
***Time-series not included in regression analyses for "rate of areal increase" and "time-to-leveling."*

Increasing proportion of soil carbon inclusion in transactional credit potential quantification (methodologies *B* to *TOT*: biomass-only to all biomass and soil carbon assumed creditable) substantially increased predicted project carbon credit potential under all scenarios (**Table 4**). However, high costs of soil compartment monitoring (USD 1,170; **Supplementary Table 2**) reduced mean predicted return-on-investment (ROI) under model-predicted and observed NR scenarios under potential accreditation methodology *BAS* to below that of biomass-only accreditation *B*. Focusing at a central USD \$9.70 Mg CO_2e potential voluntary carbon market price and a 1.5% discount rate, positive ROI was forecasted under model-predicted NR scenarios 1 and 2 under potential accreditation methodology *B* (USD \$387 and \$515, respectively) but negative ROI forecasted under methodology *BAS* (USD –\$336 and –\$64, respectively) (**Table 4**). ROI under potential accreditation methodology *TOT* was also lower than under biomass-only methodology *B* under model-predicted NR scenario 1 (USD –\$332 relative) and observed NR scenario 2 (USD –\$60 relative) (**Table 4**). The ANR scenario 3 resulted in the greatest predicted biomass and soil organic carbon sequestration and carbon credit potentials (**Table 4**). High costs of ANR (USD \$3,835.68; **Supplementary Table 2**) under scenario 3 resulted in negative mean predicted ROI under potential accreditation methodology *B* (biomass-only: economic loss of USD –\$1,905). Predicted risk of negative ROI with ANR (scenario 3) was absent under potential accreditation methodologies *BAS* and *TOT* at potential voluntary carbon market prices $\geq$ \$9.70 and a 1.5% discount rate despite soil compartment monitoring costs [all biomass and (autochthonous) soil carbon assumed creditable], where predicted ROI exceeded that under both NR scenarios *1 and 2* (e.g., USD \$431 and \$1,885 relative to scenario 1, and USD \$159 and \$1,485 relative to scenario 2, respectively) (**Table 4**). However, negative ROI was predicted under ANR scenario *3* with potential accreditation methodology *BAS* at discount rates $\geq$ 3.5% (**Table 4**). A higher carbon price of \$25.00 predicted positive ROI under all scenarios and potential accreditation methodologies. Here, forecasted ROI was similar across all scenarios with biomass-only accreditation, *B*: USD \$997 under model-derived NR scenario 1; \$1,328 under observed NR scenario 2, and; \$1,139 under ANR scenario 3 (**Table 4**). Forecasted ROI with potential accreditation methodologies BAS and TOT was, however, substantially higher under ANR scenario 3 than under NR scenarios [USD \$7,161 and \$10,910 relative to model-predicted NR scenario 1, respectively, and USD \$6,459 and \$9,878 relative to observed NR scenario 2, respectively (at 1.5% discount rate)] (**Table 4**).

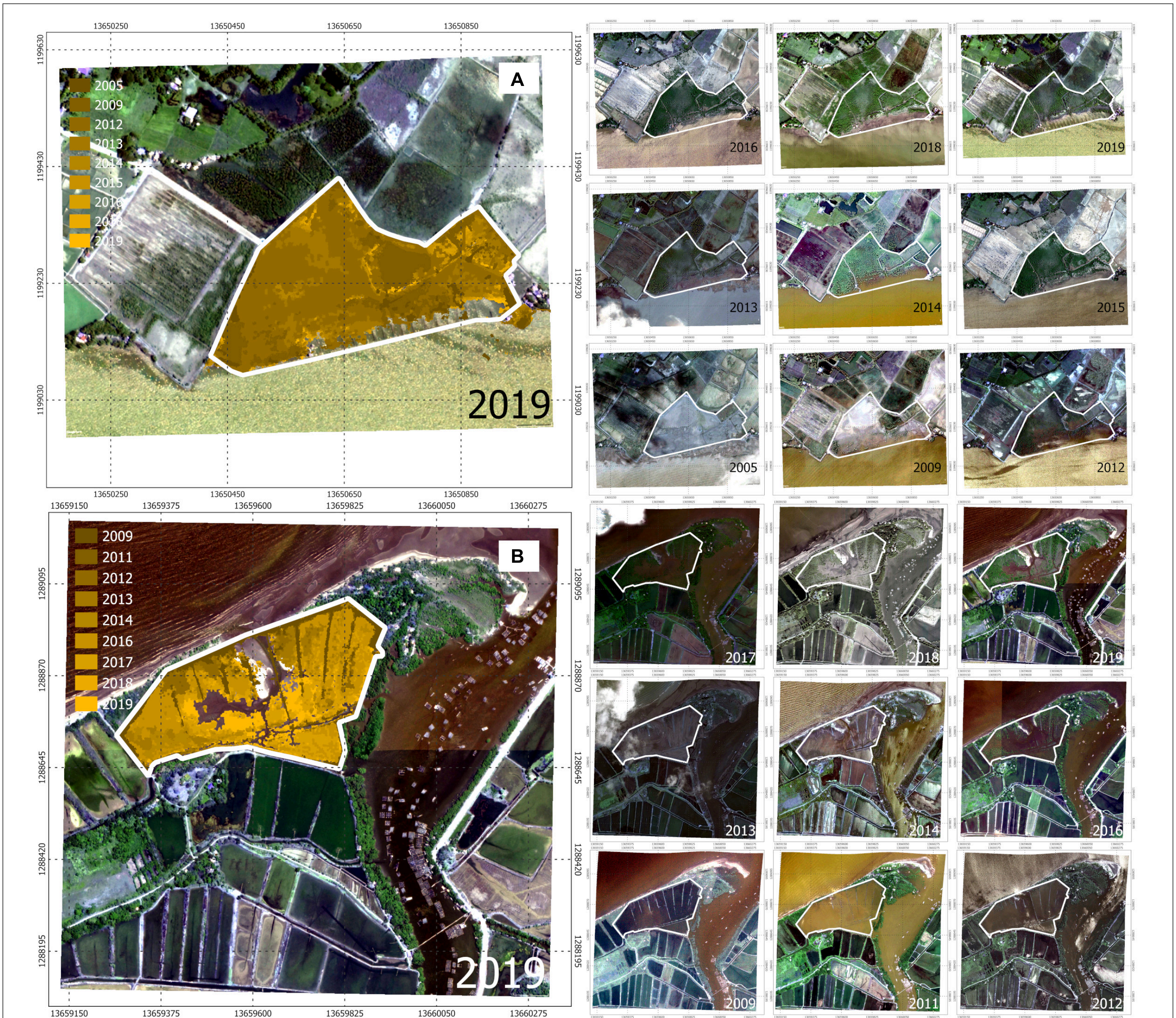

FIGURE 2 | Imagery segmentation classification of available annual Google Earth RGB imagery from abandoned pond dike breaching at **(A)** the Leganes Katunggan, Iloilo study site over 2005–2019 [natural regeneration (NR) to 2009, followed by assisted natural regeneration with planting (ANR) from 2009; see **Figures 3H,I**], and **(B)** the Roxas City, Capiz study site (NR over 2009–2019; see **Figure 3F**). See **Supplementary Figures 1A–F** for all remaining NR abandoned pond sites.

DISCUSSION

Little empirical guidance currently exists to enable mangrove rehabilitation practitioners to evaluate the potential ROI and viability of blue carbon project options. Our study provides the first to these authors' knowledge that explores drivers and variation in mangrove regeneration rates in converted coastal areas at multi-site scales, and to quantify relative ROI in potential rehabilitation-oriented blue carbon project scenario options. We observed substantial variation in mangrove NR across abandoned aquaculture ponds, with the best-fitting model explaining faster NR rates in lower-lying (pioneer low- to mid-intertidal zone species) and less exposed ponds (greater dike retention). Mangrove recolonization was substantially faster and less variable (lower risk) in our case study pond under assisted natural regeneration (ANR; planting 2009–2013) than under either observed (2005–2009) or best-fitting model-predicted NR according to site conditions (**Tables 2**, **4**). This translated to 3.7- to 5.2-fold greater carbon sequestration and 2.5- to 3.4-fold greater greenbelt regeneration (coastal protection: predicted pond mangrove coverage) over our hypothetical 10-year forecasting period with ANR. However, deducting generated potential carbon credit finance for realized project costs, ROI was low under all scenarios at current mean Afforestation/Reforestation project voluntary market carbon prices and ANR afforded a more optimal (higher ROI)

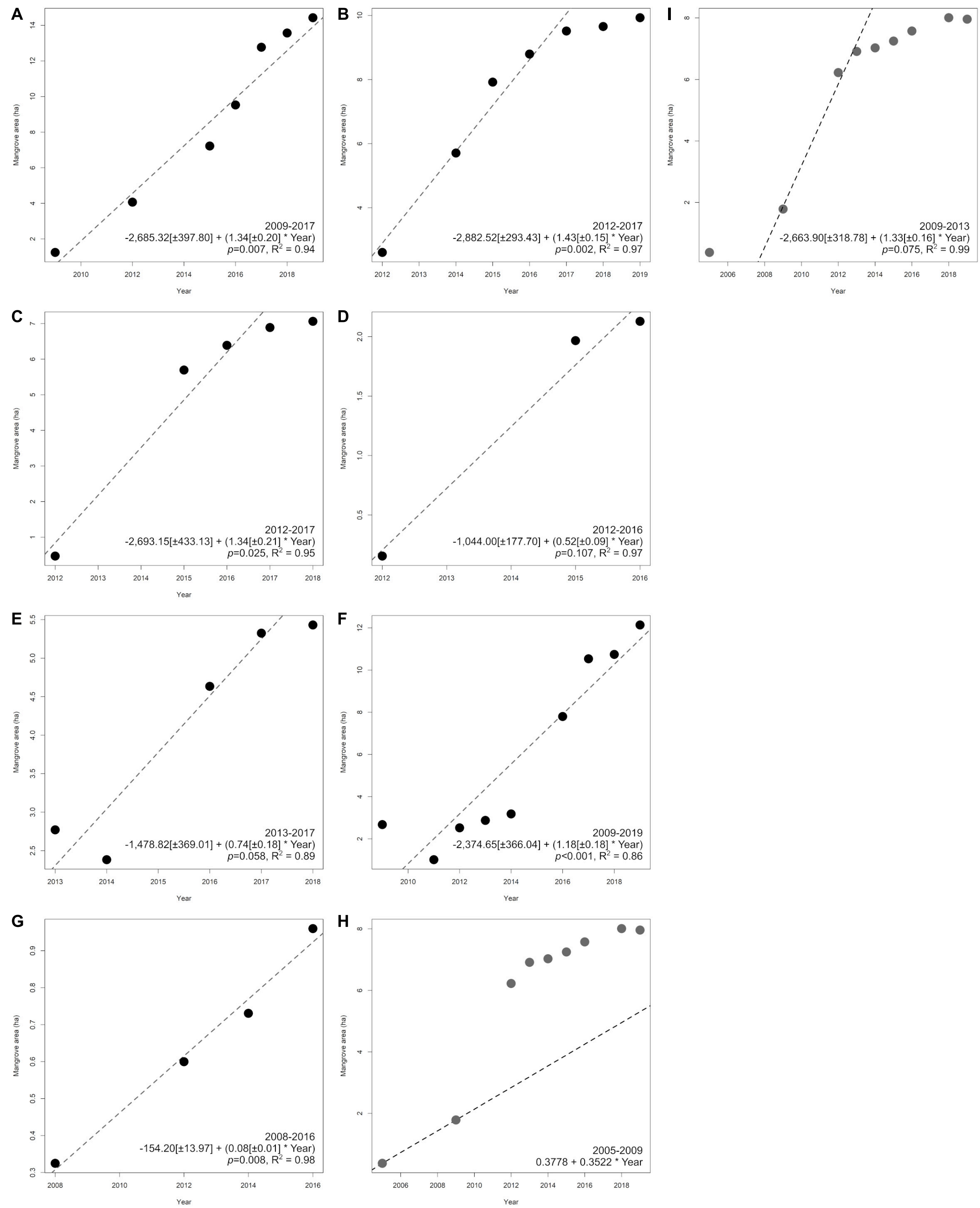

FIGURE 3 | Plots of mangrove area (ha) through classified time-series high-resolution RGB GE imagery to identified "leveling-off" of rates areal gain for all abandoned aquaculture ponds [rates in ponds **(D,F,G)** were not observed to "level-off" within the analyzed time-series]. Dashed lines show time-series linear regression models of mangrove area (ha) predicted by time (year). Naturally-regenerating abandoned ponds: **(A)** Buenavista, Guimaras; **(B)** Balagon and Napnud, Leganes; **(C)** Barotac Nuevo; **(D)** San Dionisio; **(E)** Panay; **(F)** Roxas City; **(G)** Basiao, Ivisan. Actively-rehabilitated abandoned pond (planting): **(H)** natural regeneration (NR) pre-planting (2005-2009) and **(I)** assisted natural regeneration (ANR) with planting (2009-2013) at Leganes Katunggan abandoned pond. N.B. "rate of areal increase" (ha $year^{-1}$) was calculated from a real change between the two time-points of 2005 and 2009 imagery and a constant rate applied to year 2019 for Leganes Katunggan abandoned pond in panel **(H)** (see also **Table 3**).

TABLE 3 | Top six best-fitting candidate linear models for "rate of areal increase."

Rate of areal increase (ha year^{-1})						
Explanatory variables	**AICc**	**ΔAICc**	w_i	R^2	***p*-value**	**Estimate**
Mean pond elevation (m)	*12.11*	*0.00*	*0.52*	*0.75*	*0.005*	*–0.35 ± 0.08*
Mean pond elevation (m) [β1] + *Prop. remaining pond dikes* [β2]	*13.19*	*1.08*	*0.30*	*0.91*	β1*: 0.004* β2*: 0.030*	β1*: –0.30 ± 0.06* β2*: 1.12 ± 0.37*
Var. in pond elevation (CV; %)	15.30	3.18	0.11	0.63	0.019	0.005 ± 0.002
Prop. remaining pond dikes	18.66	6.54	0.02	0.43	0.077	1.73 ± 0.81
Mean pond elevation (m) [β1] + Relative dike protection [β2]	19.78	7.67	0.01	0.80	β1: 0.007 β2: 0.332	β1: –0.36 ± 0.08 β2: –0.001 ± 0.001
Mean pond elevation (m) [β1] + log(Distance to source pop.) (km) [β2]	19.84	7.73	0.01	0.79	β1: 0.009 β2: 0.341	β1: –0.40 ± 0.10 β2: –0.06 ± 0.06
+21 further models	21.08-39.66	8.97-27.55	0.01-<0.01	0.95-0.01		

*AICc, model Akaike Information Criterion for small sample sizes; ΔAICc, model delta AICc; w_i, model Akaike weight; R^2, model multiple R-squared; Estimate, mean linear regression model slope estimate (±1 s.e.). See **Table 1** for information on explanatory variable calculation.*

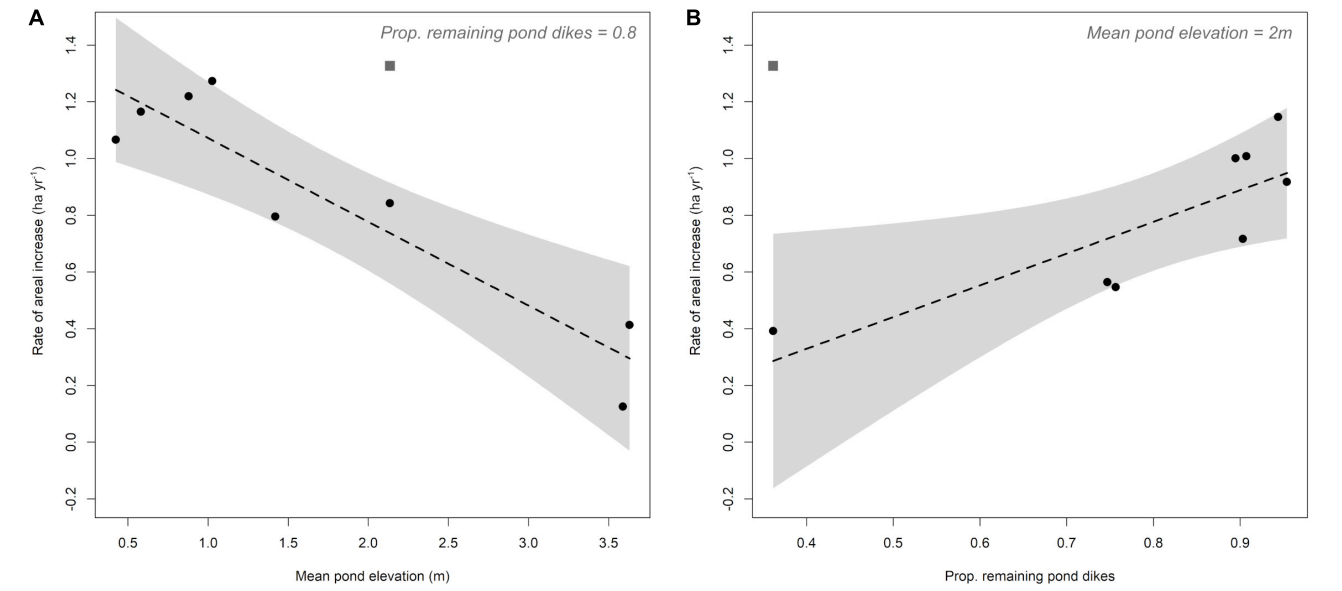

FIGURE 4 | Partial regression plots from the best-fitting linear regression model for "rate of areal increase" [mean pond SRTM DEM elevation (m; β1) + proportion of remaining pond dikes (β2)] for naturally-regenerating (NR) abandoned aquaculture ponds – black circles [model: intercept = 0.47 ± 0.35 (1 s.e.), β1 = –0.30 ± 0.06 (1 s.e.), β2 = 1.12 ± 0.37 (1 s.e.), 5 d.f., $t1$ = –5.18, $t2$ = 3.00, p = 0.002, multiple R^2 = 0.91]. **(A)** Partial regression plot of the relationship between rate of areal increase (ha year^{-1}) and mean pond SRTM DEM elevation (m), where proportion of remaining pond dikes = 0.8; **(B)** partial regression plot of the relationship between rate of areal increase and proportion of remaining pond dikes, where mean pond SRTM DEM elevation = 2 m. The dark gray square in each partial regression plot denotes the rate of areal increase observed under assisted natural regeneration [ANR (planting): 2009–2013] at the Leganes Katunggan, Iloilo abandoned pond. Gray shaded areas delimit the 95% confidence intervals of the partial regression model fits.

rehabilitation strategy over NR only where mangrove soil compartment carbon sequestration was included in potential accreditation methodologies in our case study pond site. Projected financial returns (ROI) from ANR with autochthonous soil carbon inclusion (potential accreditation methodology *BAS*) were moreover negative and similar to those under NR at higher discount rates, highlighting that neither option may be appropriate under a small-scale blue carbon project financing-only lens at current credit prices. However, where site conditions (i.e., elevation, exposure) are less optimal, our findings reveal the relative merit of ANR over slower and more variable NR to strengthen coastal protection greenbelts more quickly in the face of accelerating global climate change, which may attract additional prospective investors to inflate realized project credit payments (Plan Vivo, 2013; Beeston et al., 2020; The Blue Natural Capital Financing Facility, 2021) toward those that also substantially maximize ROI over NR alone (Turner et al., 2021). We therefore urge prospective managers to consider these context–dependencies in local conditions constraining rapid NR, as well as to identify co-benefit credit pricing opportunities to ensure positive and greater relative ROI from blue carbon project. Overall, our approach provides a means to quantify mangrove

TABLE 4 | Potential mangrove carbon sequestration profiles and voluntary market carbon credit transaction potential from model-predicted and observed regeneration rate scenarios at the Leganes Katunggan, Iloilo abandoned aquaculture pond.

		Scenario		
		1 – Model-predicted NR rate	**2 – Observed NR rate (2005-2009)**	**3 – Observed ANR rate (2009-2013)**
Total cover and sequestration	**Mangrove area (ha)**	2.84 ± 2.57 (0.28-5.41)	3.90	9.68*
	Total soil C_{org} gain (Mg)	40.65 ± 33.35 (4.23-118.36)	59.96 ± 22.69 (27.52-87.48)	227.25 ± 53.26 (129.95-420.05)
	Total biomass C_{org} gain (Mg)	20.98 ± 26.32 (0.00-119.18)	27.84 ± 25.29 (0.00-88.09)	96.11 ± 26.51 (49.19-191.93)
Methodology B	**Carbon credit potential**	44 ± 56 (0-252)	59 ± 53 (0-186)	219 ± 60 (112-437)
	ROI potential (USD)	\$100[1†] (88[1§]; 84[1¶]) **\$387[2†] (342[2§]; 326[2¶])** \$997[3†] (880[3§]; 841[3¶])	\$113[1†] (117[1§]; 112[1¶]) **\$515[2†] (454[2§]; 433[2¶])** \$1,328[3†] (1,169[3§]; 1,116[3¶])	–\$3,336[1†] (–3,393[1§]; –3,412[1¶]) **–\$1,905[2†] (–2,125[2§]; –2,198[2¶])** \$1,139[3†] (575[3§]; 385[3¶])
Methodology BAS	**Carbon credit potential**	95 ± 90 (5-346)	126 ± 73 (35-256)	579 ± 145 (317-1,105)
	ROI potential (USD)	–\$954[1†] (–979[1§]; –988[1¶]) **–\$336[2†] (–434[2§]; –466[2¶])** \$979[3†] (728[3§]; 644[3¶])	–\$884[1†] (–918[1§]; –929[1¶]) **–\$64[2†] (–195[2§]; –239[2¶])** \$1,681[3†] (1,342[3§]; 1,229[3¶])	–\$3,686[1†] (–3,839[1§]; –3,891[1¶]) **\$95[2†] (–499[2§]; –698[2¶])** \$8,140[3†] (6,610[3§]; 6,097[3¶])
Methodology TOT	**Carbon credit potential**	139 ± 123 (10-472)	185 ± 93 (67-348)	789 ± 195 (437-1,493)
	ROI potential (USD)	–\$853[1†] (–890[1§]; –902[1¶]) **\$55[2†] (–88[2§]; –136[2¶])** \$1,986[3†] (1,618[3§]; 1,495[3¶])	–\$750[1†] (–800[1§]; –816[1¶]) **\$455[2†] (262[2§]; 197[2¶])** \$3,018[3†] (2,520[3§]; 2,534[3¶])	–\$3,208[1†] (–3,418[1§]; –3,489[1¶]) **\$1,940[2†] (1,128[2§]; 856[2¶])** \$12,896[3†] (10,803[3§]; 10,102[3¶])

*Areal gain- and empirical sequestration-based carbon regeneration profiles are predicted over a 10-year timeframe from 2005 to 2015 for model-predicted (mean pond elevation + proportion of remaining pond dikes: see **Table 3** and **Figure 4**) and observed areal gain under natural regeneration (NR; scenarios 1, 2), and observed areal gain under assisted natural regeneration [ANR (planting): scenario 3]. Mean 2020 voluntary market carbon credit transaction prices (all: USD \$2.51 Mg CO_2e; Afforestation/Reforestation: USD \$9.70 Mg CO_2e) and potential credit transaction price with increased demand (USD \$25.00) and discount rates of 1.50, 3.50, and 4.25% are applied (see section "Materials and Methods") to calculate transactional carbon credit potential value under three potential accreditation methodologies: B: only biomass CO_2e emissions reduction; BAS: biomass CO_2e and autochthonous soil organic carbon (53.28 and 63.71% for NR and ANR scenarios, respectively) emissions reduction, and; TOT: biomass CO_2e and all soil organic carbon emissions reduction. Return-on-investment (ROI) in potential accreditation methodologies BAS and TOT is deducted for costs of soil carbon sequestration monitoring (USD 1,170), and ROI in ANR scenario 3 is also deducted for observed costs of ANR (wilding collection, nursery rearing and out-planting) (USD 3,835.68) (**Supplementary Table 2**).*

NR, natural regeneration; ANR, assisted natural regeneration; ROI, return-on-investment.

**Abandoned pond (total area 9.68 ha) completely mangrove-vegetated.*

[1]Price at USD \$2.51 Mg CO_2e (see section "Materials and Methods").

[2]Price at USD \$9.70 Mg CO_2e (see section "Materials and Methods").

[3]Price at USD \$25.00 Mg CO_2e (see section "Materials and Methods").

[†]Discount rate of 1.50% applied (see section "Materials and Methods").

[§]Discount rate of 3.50% applied (see section "Materials and Methods").

[¶]Discount rate of 4.25% applied (see section "Materials and Methods").

Bold figures show main findings discussed in the manuscript.

regeneration and its drivers in converted lands across relevant (i.e., multi-site) geographic contexts and utilize these to evaluate rehabilitation-oriented blue carbon project scenarios that take in to account these enabling processes in local conditions, and "plug-in" forecasted project-specific costs.

While, as mangrove regeneration rates for our Leganes Katunggan, Iloilo case study were greatest under the forecasted ANR scenario (1.327 vs. 0.247 ha year^{-1} in best-fitting model-derived NR scenario 1 and 0.352 ha year^{-1} in observed NR scenario 2), we observed wide variability in NR rates across the seven studied NR abandoned ponds that were in three cases greater than the case study ANR scenario (**Table 2**). Our employed methodology and an absence of ground truth data meant we were unable to conduct time-series classification accuracy assessment. However, we are confident that our mapping approach combining manual visual classification of high-resolution imagery segmentation produced high accuracy (with low variability due to a single visual classifier analyzing each image) in estimated NR rates, and therefore that the rates quantified reflect context-dependence in NR drivers across sites. Depending on pond-specific conditions, ANR could still enhance some high NR rates observed by enhancing stem density, soil stability and inter-individual facilitation (Huxham et al., 2010) even under their apparently optimal local conditions. Our study could identify only one case study pond in which multiple regeneration processes (early NR followed by ANR: Primavera et al., 2012b) had occurred in the region, and further empirical and/or experimental study of mangrove NR vs. ANR rates at ponds with varying site-specific conditions is needed to identify the conditions at which ANR becomes redundant to background NR rates (see also Wodehouse and Rayment, 2019). Further mapping and quantification of NR across varied and more numerous abandoned ponds would also be beneficial in refining driver models: only eight abandoned ponds were identified here and we did not observe wide ranges of propagule supply within our model dataset (**Table 2**). We observed high NR rates in ponds with lower intertidal elevation (~0.4–0.1 m a.m.s.l.) and low exposure (~90% sea-facing dike retention) (**Table 2**). However, it is possible that the high NR rates observed at some abandoned ponds (**Table 2**) are driven by other known NR-influencing factors such as propagule supply (Balke et al., 2011; Lewis and Brown, 2014), where low sample size and variation in our dataset did not enable detection of a relationship, contrary to our hypothesis (**Table 1**). Interactions with factors that it was not possible to include in our landscape-level analysis, such as substrate type and salinity, could also have further driven variation in observed NR rates. We searched the high pond abandonment West Visayas coastal zone (∼31% of production from the Philippines total 239,323 ha of coastal aquaculture ponds: Department of Agriculture of the Philippines – Bureau of Fisheries and Aquatic Resources, 2020) exhaustively for study ponds, but a lack of high resolution imagery pre-2009 resulting in non-inclusion of many NR ponds for which we could not establish time of NR initiation. Future quantification of pond abandonment and NR could be evaluated against, and potentially supplemented by, mapping with longer-term moderate resolution imagery (e.g., Landsat) (Duncan et al., 2018; Baloloy et al., 2020); however the fine-scale of mangrove regeneration (i.e., individual tree establishment) may reduce the utility of these methods. Going forward, the generation of greater quantities of high resolution RGB imagery with existing and to-be-deployed satellite constellations will substantially enhance our ability to map and monitor newly abandoned coastal areas and refine NR driver models (Curnick et al., 2021).

Higher regeneration rates under ANR translated to substantially greater carbon sequestration and credit potential than under slower and more variable NR (**Table 4**). However, contrary to our predictions, application of blue carbon Verified Carbon Standard accreditation protocols (Verra, 2020b) and deduction of project costs (**Supplementary Table 2**) at current mean Afforestation/Reforestation project voluntary market carbon prices (USD \$9.70 Mg CO_2e; Forest Trends' Ecosystem Marketplace, 2021), resulted in ROI not only lower than NR scenarios but negative (net losses of USD \$1,905 to \$2,198) under the ANR scenario at Leganes Katunggan under biomass-only potential accreditation methodology (*B*). Biomass carbon sequestration at the case study ANR abandoned pond with sub-optimal conditions (elevation, exposure) is at the lower end of empirical estimates for rehabilitating mangroves (Sasmito et al., 2019), and biomass-only carbon emissions reductions may be sufficient to return positive ROI relative to ANR project costs in different contexts. However, our case study illustrates that biomass-only accreditation, accounting for only a small proportion of full emissions reduction portfolios, returns lower relative ROI in ANR rehabilitation-only blue carbon project scenarios. Thus NR may be the appropriate strategy under biomass-only accreditation in small-scale potential blue carbon projects at current mean credit prices (Afforestation/Reforestation projects). Relative ROI under ANR increased to marginally to substantially greater with methodologies with increased inclusion of soil compartment emissions reductions at a 1.5% discount rate (**Table 4**); yet, projected ROI under ANR with autochthonous soil carbon inclusion (methodology *BAS*) was negative and similar to that projected under NR at higher discount rates $\geq$ 3.5%. Thus neither NR nor ANR rehabilitation option may be appropriate under a small-scale blue carbon project financing-only lens with soil carbon inclusion at current credit prices, where soil compartment monitoring and verification inhibits ROI, unless additional co-benefits can drive higher credit pricing (see Plan Vivo, 2013; Mikoko Pamoja, 2020). Importantly, ANR was the only rehabilitation scenario that forecasted full mangrove cover within the 10-year time frame at the 9.58 ha Leganes Katunggan case study pond (upper NR estimate: 3.90 ha cover; **Table 4**). ANR therefore establishes wider greenbelts more quickly in abandoned ponds with sub-optimal NR conditions (i.e., elevation and exposure), which is critical for surge reduction in typhoon-prone areas and where national greenbelt mandates are a long way from being met (Primavera et al., 2012a). This additional CCMA benefit could furthermore strengthen ANR blue carbon projects with accreditors enabling higher payments for "bundled" co-benefits (e.g., Plan Vivo, 2013), allowing them to further recoup implementation costs from enhanced coastal resilience. This could see the relative ROI of ANR approaches outweighing that of NR alone at levels similar to those reported here at USD \$25.00 Mg CO_2e prices: ∼4.8- to 8.3-fold greater relative ROI under ANR with autochthonous soil carbon inclusion (*BAS* methodology) (**Table 4**). ANR project scenarios moreover have the potential to be designed to further enhance site-specific carbon sequestration (Bai et al., 2021; Rahman et al., 2021) and associated co-benefits (coastal protection: Duncan et al., *unpublished data*) through introducing greater species and taxonomic diversity at early rehabilitation stages.

The limited number of and wide variation in empirical studies on carbon sequestration in NR abandoned ponds in SE Asia furthermore drove high uncertainty in emissions reductions under NR scenarios (see section "Materials and Methods"; **Table 4**). Further on-ground research to quantify carbon sequestration in NR rehabilitating mangroves and their drivers may serve to reduce uncertainty in potential emissions reductions and therefore potential buyer confidence in *ex ante* NR-generated blue carbon credits. At present, this uncertainty, combined with uncertainty in model-derived NR rates (**Figure 4**), triggered deductions in project scenario emissions under NR for all potential accreditation methodologies (5.01–11.28% deductions). Coupled with increased natural risk from soil erosion at the exposed site (5% additional non-permanence buffer applied; Verra, 2019), this could sum to substantial perceived project risk to credit-buyers. Instead, blue carbon project developers could seek to increase potential project viability in terms of credit sales by prioritizing reduced uncertainty (ANR rehabilitation strategies) over slow and variable NR alone. Here, a combination of perceived risk reduction (i.e., lower discounting) alongside faster rates of mangrove greenbelt regeneration may attract substantial interest from other relevant industry investors to drive up credit pricing and realized ROI (see Beeston et al., 2020; Sumaila et al., 2021). This reduction of credit-buyers' perceived risk with ANR rehabilitation efforts may also be combined with less costly conservation actions (i.e., avoided deforestation/degradation) to maximize both pricing and carbon sequestration in larger-scale blue carbon project planning (see Mikoko Pamoja, 2020; Conservation International, 2021; The Blue Natural Capital Financing Facility, 2021).

There remain other substantial practical considerations that should be factored in to the development of blue carbon projects beyond those that could be explored in our case study. First, we conducted case study-specific non-permanence risk assessment (Verra, 2019), where low exposure in coastal configuration (adjacent island) reduces risk from highly frequent typhoon events in the country, resulting in moderate non-permanence credit buffers (30–35%). Natural disaster risk, and others such as political risk, in other potential sites may exceed certification thresholds (Verra, 2019) or mean that non-permanence buffers drastically restrict potential ROI without active intervention to reduce stochastic threats to permanence in susceptible regions. Second, our study considered low-intensity, sea-facing abandoned ponds for which the need for physical rehabilitation interventions to reinstate hydrology are minimal (natural storm-driven dike breaching: Primavera et al., 2014). On-site physical intervention costs on initiation to make projects additional (Verra, 2020b) (e.g., dike breaching in less exposed sites and/or additional hydrological interventions) for NR rehabilitated-oriented projects in other contexts may be substantial (Su et al., 2021). With potential physical intervention costs applied, our hypothetical 9.68 ha) case study project would not be financially viable under either rehabilitation strategy at current voluntary carbon market prices (USD \$9.70 Mg $CO2e^{-1}$: Forest Trends' Ecosystem Marketplace, 2021; see also Thompson et al., 2014). Furthermore, costs to both NR and ANR blue carbon project strategies may increase substantially where community engagement for project activities is low or absent (Primavera et al., 2012b). Variation in such project-specific biophysical costs may tip the balance of relative ROI between rehabilitation strategy options (NR vs. ANR), and high costs may in some cases make all options prohibitively expensive. Most importantly, our quantification of project costs is simplistic, focusing only on biophysical costs. In reality, any blue carbon project requires large budgets for certification standard fees (registration and verification) and any technical capacity not held by the proponent organization. These costs can be substantial (∼USD 40,000 for our hypothetical case study: Verra, 2020c), and realization of an operational blue carbon project, let alone sufficient financial ROI, is often not possible without early impact or seed investment from a non-credit purchasing body (see Mikoko Pamoja, 2020; The Blue Natural Capital Financing Facility, 2021). Put together, these cost and risk context-dependencies invite application and extension of the approaches developed herein to other and wider mangrove blue carbon rehabilitation contexts, urge caution to prospective blue carbon project developers and conduct of driver analyses such as performed here to evaluate real-world project scenario options.

With the arrival of the approved blue carbon Verified Carbon Standard methodology (Verra, 2020a,b) we are likely to see a rapid increase in demand for blue carbon credits in the coming years. To maximize blue carbon's potential to advance global CCMA efforts and unlock sustainable conservation financing, it will now be imperative to evaluate the economic viability of potential blue carbon project scenario options prior to embarking on lengthy and costly project registration processes. With this study, we provide an approach to quantify the ROI and viability of potential rehabilitation-oriented blue carbon project scenarios, using projected biophysical project costs and multi-site scale ecological models to predict rates of transactional credit generation against likely biophysical cost. We employ a case study in which biophysical rehabilitation intervention requirements are minimal, revealing potentially substantial carbon sequestration and reduced uncertainty in such moderate-scale ANR rehabilitation blue carbon projects now able to incorporate significant soil carbon compartments. While ANR rehabilitation costs in other contexts are highly variable and can be considerable (Su et al., 2021), our case study is representative of coastal mangrove rehabilitation opportunities across large parts of SE Asia where coastal aquaculture and pond abandonment is extensive (Richards and Friess, 2016; Goldberg et al., 2020). We thus highlight a potential opportunity for additional ANR blue carbon projects in SE Asian countries such as the Philippines, where much former mangrove extent has already been cleared, rates of loss are decreasing (Spalding et al., 2010; Goldberg et al., 2020), national mandates forbid further unplanned deforestation (Primavera et al., 2012a), and coastal greenbelt rehabilitation for CCMA is high on the political agenda. Instead, ANR rehabilitation-oriented blue carbon strategies in abandoned, sea-facing ponds under non-private ownership, and for which tenure reversal and political will strengthening are the limiting factors to additionality, represent a widespread, comparatively low-cost CCMA opportunity. Generating the co-benefit of reducing perceived risk to credit-buyers and more rapidly strengthening coastal greenbelts in typhoon-prone regions, such a strategy could moreover attract higher credit pricing and advance national CCMA mitigation strategies, where ANR strategies could have substantially greater (∼4.8- to 8.3-fold) relative ROI than NR alone. It is important to stress that any implemented ANR rehabilitation approach for blue carbon credit generation should employ scientifically-founded guiding principles, and that traditionally-performed widespread mangrove planting (inappropriate species, intertidal locations) is highly unlikely to produce successful blue carbon projects (Primavera et al., 2012b; Lee et al., 2019) or positive financial returns (as yet unstudied). Approaches such as those developed herein can guide optimal site-specific rehabilitation implementation options within such a strategy to maximize potential sustainable conservation financing.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author.

AUTHOR CONTRIBUTIONS

CD, JP, and HK conceived the research ideas. CD collected the data, performed analyses, and wrote the first draft of the manuscript. All authors contributed critically to wrote the final draft of the manuscript and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/ffgc.2021.775341/full#supplementary-material

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Ecosystem Services Assessment for the Conservation of Mangroves in French Guiana Using Fuzzy Cognitive Mapping

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In 2016, the French government adopted a law for biodiversity, setting an objective of protecting 55,000 hectares of mangroves. This objective is particularly important to French Guiana, which shelters almost 60% of French mangrove ecosystems, and where mangroves occupy three quarters of the coastline. The coast of French Guiana is also where issues associated with demographic and economic dynamics concentrate. There is thus a need to plan for an economic development that is compatible with the objective of protecting mangrove ecosystems. Ecosystem services (ES) assessment can support such decision-making, informing on the costs and benefits associated with alternative mangrove conservation strategies. While the many services provided by mangrove ecosystems are well documented worldwide, the extent to which these can be encountered in the specific case of French Guiana is currently only very partially known. Relying on the Fuzzy cognitive mapping (FCM) approach, we collected and compared the perception of multiple and heterogeneous groups of stakeholders, of the functioning of the mangrove social-ecological system at the scale of French Guiana. Results, allow to identify mangroves ES and threats particularly influenced by the high sedimentary dynamism of the shoreline. This generates two distinct components of the mangrove social-ecological system: mud banks where ecosystem services are spatially and temporally unstable, and associated with perceived constraints for key coastal activities, and estuarine mangroves where the ecosystem services usually described in the literature on mangroves can be found. Disservices associated with mangrove ecosystems were also identified as a key interaction. This can inform the research needs that should support sustainable development trajectories, fully accounting for the protection of French Guianese mangrove ecosystems.

Keywords: French Guiana, stakeholders perceptions, marine ecosystem services, mangrove forest, socio-ecosystem sustainability

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INTRODUCTION

Despite their importance, mangroves are disappearing at a global rate of 1–2% per year (Spalding et al., 2010) and 20–35% have been lost in the last 50 years (Polidoro et al., 2010). The main threats to mangroves are climate change (Gilman et al., 2008; Lovelock et al., 2015; Schuerch et al., 2018); land-use conversion to agriculture and aquaculture (Thomas et al., 2017;

Goldberg et al., 2020); and pollution (Maiti and Chowdhury, 2013). These threats have led to the mobilization of the international community for mangrove conservation (Sandilyan and Kathiresan, 2012; Friess et al., 2016).

Within its overseas territories, France shelters more than 80,000 ha of mangrove (Trégarot et al., 2021). In 2016, in line with its international commitments, the French government adopted an ambitious law to protect 55,000 hectares of mangroves, with the mangrove of French Guiana at the forefront of this action.

In French Guiana, mangrove forests, that occupy around 75% of the coastline, are in a relatively good state, largely un-impacted by urbanization and residential development (Trégarot et al., 2021). This situation strongly contrasts with the neighboring countries, notably Suriname and Guyana, where mangroves have been severely impacted by urbanization (Anthony and Gratiot, 2012). However, demographic projections estimate an increase of 75% of population by 2050 (INSEE, 2019). With a population predominantly located on the coast (Zouari, 2015), urbanization pressure is expected to increase as a result of both residential demand and infrastructure development (e.g., recently with the construction of a power plant; Autorité Environnementale, 2019). In addition, there is currently the implementation of a planning strategy for the maritime economy as a development opportunity for economic growth, that will also require the development of dedicated infrastructures, directly on the coast (e.g., port; CEREMA, 2016). In this context, the conservation of mangroves is likely to become a challenge for decision-makers requiring the assessment of associated trade-offs. Understanding the importance of mangroves to society is thus necessary, to support the development and implementation of informed conservation policies.

The concept of ecosystem services (ES) was introduced to account for interdependencies between human societies and ecosystems (Daily, 1997; MEA, 2005). By enabling the identification of these interdependencies, its application can inform on the trade-offs between economic development and biodiversity conservation (Carpenter et al., 2009).

Recently, France conducted a national assessment of the state of marine ecosystem and ES[1] that showed a lack of information regarding ES in its overseas territories, including French Guiana, as compared to mainland France (Mongruel et al., 2018). Regarding mangroves, information remains relatively scarce, and not all ES are equally well documented. In comparison, there is an extensive literature on the ES they provide at the global level, which identifies a range of services provided by these ecosystems, as well as interactions between these services.

Firstly, mangroves worldwide deliver many provisioning services that are essential for local and national economies (Rönnbäck, 1999). They support commercial, recreational and subsistence fisheries (e.g., Manson et al., 2005; Aburto-Oropeza et al., 2008). For example, a positive statistical relationship has been identified between catches of fish or shrimp and mangrove surface area (Carrasquilla-Henao and Juanes, 2017). Shellfish gathering can also occur directly in the mangrove (Treviño and Murillo-Sandoval, 2021). Mangroves have also been shown to sustain shrimp aquaculture (e.g., Truong and Do, 2018). However, the intensive conversion of mangroves into aquaculture farms is currently one of the main threats to mangroves in many countries, and may not be compatible with their importance in sustaining fisheries (Naylor et al., 2000). Among the other products provided by mangroves, the harvesting of wood for construction, combustible or artisanal products has also been highlighted (e.g., Walters, 2005; Bosire et al., 2008).

Secondly, mangroves have been shown to provide cultural services, although the importance of these services is less well documented (Himes-Cornell et al., 2018). Mangroves can support nature-based recreational activities that include diving, bird watching, hiking and recreational fishing (Van Oudenhoven et al., 2015), contributing to tourism development (Spalding and Parrett, 2019). Mangroves can also support the production of knowledge for research and education (Owuor et al., 2019). Mangroves are also associated with more immaterial values, where coastal communities have developed symbolic relationships with the mangrove forest (de Souza Queiroz et al., 2017).

Thirdly, mangroves have been shown to provide regulatory services. Mangroves act as a buffer between the land and the sea, significantly attenuating the energy of wind-generated surface waves (Massel et al., 1999) and protecting the coastline from tropical storms (Ouyang et al., 2018; Hochard et al., 2019). Mangroves can also play a role in regulating the impacts of human activities on water quality, studies showing the ability of mangrove ecosystems to reduce nutrient loads (Xiao et al., 2018; Adame et al., 2019) and chemical concentrations (MacFarlane et al., 2007; Kulkarni et al., 2018) in coastal water. Mangroves have also been shown to support climate regulation, through carbon sequestration (Bouillon, 2011; Duarte et al., 2013; Atwood et al., 2017). Mangroves' stocks of carbon are distributed between aboveground biomass, belowground biomass and soil (Walcker et al., 2015).

When relying on the concept of ES to understand the relationships between societies and ecosystems, studies often fail in addressing the importance of disservices (Blanco et al., 2019). Historically, mangroves were more commonly considered as a reservoir of disease such as malaria by nineteenth century explorers that led to global drainage operations (Friess, 2016). Nowadays, mangroves continue to receive negative press that can severely undermine conservation efforts (Dahdouh-Guebas et al., 2020).

The objective of this project was to begin bridging this gap between the state of knowledge at international and French Guiana levels, by developing a first comprehensive assessment of the ES provided by mangroves in this territory.

There is currently no consensus regarding the way ES assessment should be structured (Schröter et al., 2014). An important body of research has focused on the monetary valuation of the benefits that humans derive from ecosystems. Monetary values reflect the social importance of ES and are considered by many as a prerequisite for better management decision-making (TEEB, 2010; Costanza et al., 2017). However,

[1]The EFESE project for French Assessment of Ecosystem and Ecosystem Services was divided in several reports according to types of ecosystems: agricultural, urban, mountainous, wetlands, forested and marine and coastal. Mangrove ecosystems, as transitional ecosystems, were included in marine and coastal ecosystems.

the monetary valuation of ES is still subject to criticism, due to methodological and theoretical controversies (Farley, 2012; Muradian and Gómez-Baggethun, 2021) or to their operationalization (Marre and Billé, 2019).

In this paper, we rely on another facet of ES assessment which stems directly from the need to support policy-making regarding biodiversity conservation via improved understanding of the potential impacts of alternative conservation strategies (Armsworth et al., 2007). In this perspective, ES assessments take root in the contribution of systems sciences to the understanding of socio-ecological systems (Braat and de Groot, 2012). This approach is also strongly influenced by the concepts of biological conservation, for which the imperative of providing policy answers to the biodiversity crisis imply adopting holistic multidisciplinary approaches, that also include stakeholder knowledge. Following this path, ES assessment constitutes a boundary object allowing various stakeholders to share their representation of the world based on a common framework (Steger et al., 2018).

With this in mind, and given the lack of prior studies of mangrove ES in French Guiana, we relied on the expertise of French Guiana stakeholders to develop the first holistic assessment of ES associated with Guianese mangrove ecosystems. We used fuzzy cognitive mapping (FCM) as an integrated research tool to assess how stakeholders perceive the entire bundle of multiple and interconnected ES within the mangrove socio-ecosystem. FCM is a semi-quantitative modeling tool that is useful to analyze and compare stakeholders' knowledge of a socio-ecosystem (Özesmi and Özesmi, 2004; Gray et al., 2014; Bosma et al., 2017). In addition, FCM are easy to use in participatory research settings (van Vliet et al., 2010). We collected stakeholders' perception using a combination of interviews and workshops.

First, we provide a synthetic description of mangroves in French Guiana and a presentation of the FCM methodology and how this was used to develop a holistic representation of the Guianese mangrove socio-ecological system. This representation is then described, taking into account the qualitative information collected as part of focus groups where it was presented to stakeholders. The article then discusses the implications of this representation for mangrove conservation policy in French Guiana, and concludes.

MATERIALS AND METHODS

Presentation of the Case Study

French Guiana is an overseas department of France, bordered by Brazil in the south and east and by Surinam in the west, with a land area of 83,534 km^2 and a coastline of 320 km in length. Its population of around 275,000 inhabitants in 2016 is mainly concentrated within 10–30 km-wide coastal strip (Zouari, 2015). The main city is Cayenne where the main transport infrastructures of the territory are located (international airport, main commercial port) and which shelters half of the Guianese population. The two other main populated areas are Kourou where the Guianese Spatial Center is located and the estuary of Maroni river that is characterized by the highest population growth.

It is located in a equatorial climate and has all the necessary characteristics for mangrove colonization and growth, with air temperature fluctuating between 26 and 30°C and rainfall ranging from 2,500 to 3,000 mm.yr^{-1} (Marchand, 2017; Walcker et al., 2018). Mangroves in French Guiana occupy almost 75% of the coastline (Walcker et al., 2015).

French Guiana's coastline is characterized by the dynamics of its coastline, that is deeply affected by the Amazon River. Sediments eroded in the Andes are transported along the Amazonian basin down to the river mouth (Martinez et al., 2009). There, under the influence of tides, waves and currents, accumulated sediments form individual mud-banks typically extending 10–60 km along the coastline, 20–30 km offshore and 5 m thick, that move along the coast toward the Orinoco river (Venezuela) at a rate of 1.5–3.5 km/yr (Gardel and Gratiot, 2005). This dynamic geomorphology offers uncommon conditions for coastal mangroves (Walcker et al., 2018). During their formation, mud-banks are colonized by propagules carried by the tides; with the arrival of new seeds the mangrove forest accumulates in successive strips of even-aged stands. During the erosion phase, mangroves are swept away starting with the youngest forest stands, until the erosion stops or until the mangrove locally disappears with its substrate. Coastal mangroves, exposed to mud-banks migration, are mostly dominated by *Avicennia germinens* that are more effective in rapidly colonizing and developing on such an unstable and stressful substrate (Fromard et al., 2004).

Mangrove succession in French Guiana is also well described (Fromard et al., 1998) starting with pioneer mangroves that accumulate biomass while maturing. Mangroves are dominated by *Laguncularia racemosa* and *A. germinens*, and depending on environmental conditions, sediment can accumulate and the landward mangrove forest can turn into savannah. Under stronger riverine influence, mangrove species association also include *Rhizophora* spp. and can evolve into marshy forest, depending on sedimentary conditions. Excess sedimentation can also suffocate mangrove trees, leading to dead mangrove areas, that can enter a new cycle of colonization, if conditions are favorable. Such dead mangrove areas are a characteristic feature of the Guianese coast (Fromard et al., 1998). In a nutshell, mangrove in French Guiana are not homogenous and an important distinction exists, between coastal mangroves that are exposed to mud-bank migration and estuarine mangroves located under riverine influence.

Fuzzy Cognitive Mapping

Presentation of the Method

FCM is an integrated research tool that has been developed to assess and compare expert knowledge (Özesmi and Özesmi, 2004). The approach presents many advantages relevant to our research question, notably the ability to model system relationships where scientific information is limited but expert and stakeholder knowledge is available, and the ability to deal with variables that may not be well-defined. As a consequence, FCM captures all knowledge, including individual misconceptions or biases. However, this can be reduced by the possibility of combining individual answers, thus limiting the uncertainty associated with individual responses (Özesmi and Özesmi, 2004). FCM can either be used to capture the knowledge of experts (e.g., Hobbs et al., 2002) as well as non-experts, including local stakeholders (e.g., Gray et al., 2015). In this perspective, it has successfully been implemented in both social and ecological research (Teixeira et al., 2018).

A cognitive map can be defined as "a qualitative model of how a given system operates" (Özesmi and Özesmi, 2004, p. 44). It offers a graphical representation of the relationships between the key variables of the system. Variables can designate physical quantities that can be measured (e.g., a number or biomass of fish) or more abstract concepts (e.g., heritage value). Drawing a cognitive map thus implies (i) selecting the important variables that affect a system and (ii) establishing the causal relationship among these variables with a number between −1 (negative effect) and 1 (positive effect). It is the application of fuzzy causal functions to measure the connections between variables, relying on real numbers between [−1; 1] rather than integers, that turn cognitive maps into fuzzy cognitive maps (FCMs). Cognitive maps have the advantage of being concise, allowing stakeholders, including decision-makers, to capture the complexity of a system at a single glance.

Once individual FCMs are collected, it is common to aggregate them in order to form a social map (Özesmi and Özesmi, 2004), with the following steps: (1) an augmented matrix including the variables from all the individual maps is created; (2) all the individual FCMs are coded into the augmented matrix; (3) individual maps are aggregated using matrix addition resulting in a social map. It is also possible to normalize the matrix values of this social map by the number of cognitive maps underlying them, to obtain scores between −1 and +1.

After aggregation, the social map contains all the variables that have been identified by individuals. At this stage, it may be necessary to condensate the social map to avoid too many variables and connections (Özesmi and Özesmi, 2004). Condensation is the action of replacing a part of the social map with a single variable. Condensation may follow a quantitative logic—maintaining the strongest relations—or a qualitative logic—merging variables when they can be united under a larger encompassing variable. When replacing a group of variables, connections from merged variables to other variables are maintained.

The construction of social maps is based on the rationale that an assessment by many experts with diverse visions and perspectives will have greater relevance than one relying on a single expert versed in all aspects of the problem. In this work, we refer to two types of social maps: "stakeholder maps" when the social map is obtained from the aggregation of individual FCMs within a particular stakeholder group, and "community maps," when the social map is obtained from the aggregation of all the individual FCMs, across stakeholder groups.

Sampling Design and Fuzzy Cognitive Mapping Construction

There is no constraint regarding the way FCMs should be built: some researchers choose to draw individual FCMs during a face-to-face interview and then combine individual maps in a social map (Bosma et al., 2017) while others prefer drawing single FCMs during a workshop that gathers the targeted experts (Gray et al., 2015). In this research, we implemented an original methodology for developing FCMs by combining both individual interviews, and group workshops as in Gourguet et al. (2021). This process is similar to the logic of the Delphi process, where experts are asked to express their judgment several times on the same subject with the possibility to re-evaluate their judgment at each round, based on the aggregate results of the previous round. Such a process is particularly useful to find consensus on complex matters (Rowe and Wright, 1999), and thus seemed relevant to the objective of our study, which was to mobilize stakeholder knowledge to develop a holistic representation of the functioning of mangrove socio-ecosystems, that can assist in identifying key conservation levers and obstacles.

A stakeholder is usually defined as a person who affects or is affected by a decision or action (Reed et al., 2009). Given the size of the territory studied, the number of stakeholders to consider is very large. We thus narrowed our scope to expert stakeholders, i.e., stakeholders with extensive knowledge or skills on the subject of study, based on research, experience or occupation in a particular field related to mangroves. We followed the Campagne and Roche (2018) approach for expert selection. We first identified key stakeholders closely involved in the conservation of mangroves in French Guiana and followed their recommendations of additional experts to contact, resulting in a list of 29 experts from four categories: scientists, managers, conservationists and economic actors. The steps of the consultation process are presented hereafter and summarized in **Table 1**.

The first step of the work consisted in the face-to-face interviews and creation of individual FCMs. Between March and July 2019, we conducted individual interviews with the 29 experts. Interviews were divided into two parts: (i) firstly, a semi-structured interview questioned experts on their activity and its links with mangroves and (ii) secondly, the drawing of FCMs. We favored face-to-face interviews over workshops in the first phase to avoid the risk of answers based on conformity and group pressures (Woudenberg, 1991). Because of logistical difficulties (remoteness or transport difficulties in French Guiana) some experts were contacted by phone or videoconference. In those cases, it was not possible to draw the FCM, as this requires sharing the visual conception of the map while it is being developed. In the end, 19 FCMs were collected. We then homogenized the terms used across FCMs when there was no ambiguity that experts were speaking of the same variables. We used these homogenized variables to combine the individual FCMs and obtain social maps, merging the variables and summing the connections between the same variables.

TABLE 1 | Steps of the expert consultation process and links with the successive social maps produced.

Step	Activity	Material	Social maps	Number of variables
1	Face to face interviews with experts	29 interviews	Social map #1	89
2	Reduction of the number of variables	19 individual FCMs	Social map #2	29
3	Workshops with experts from each category	4 workshops	Social map #3	30

The second step of our consultation process consisted in the condensation of the community map. To do so, we applied three types of rules following three questions:

(1) Can the variables be grouped under a superior key concept following a qualitative logic? We notably relied on the framework of ecosystem services (e.g., naturalistic observation, hiking and visits by boat were merged under the broader concept of recreational activities) or factors of change from global biophysical assessments (e.g., sea level rise, drought and acceleration of mudflat migration, under the broader concept of climate change). In this case the value of the connection between group variables was divided by the number of grouped variables.

(2) Can a succession of variables be grouped, as they describe a single process? This follows a more quantitative line of reasoning, where logical chains are shortened (e.g., the chain Mangrove - > Wood production - > Handicraft - > Wooden articles is replaced by Mangrove - > Handicraft). In this case the value of the connection from deleted variables with other variables is preserved.

(3) Can we delete isolated variables? Variables that were mentioned by only one expert and that could not be grouped were removed.

All the changes involved in the condensation process were recorded, in order to be able to explain these to stakeholders in the following step.

The third step of our consultation consisted in organizing workshops by groups of stakeholders to discuss their stakeholder FCMs and identify consensual and/or conflicting views on these maps. Given the size of the mangrove socio-ecosystem in French Guiana and the complexity of the question we aimed to address, the number of maps we obtained may be considered low. Combining individual interviews with workshops offered a means to increase consistency in answers (Singh et al., 2017). We organized 4 workshops in February 2021, each workshop was open to every expert that was interviewed, whether they had drawn FCMs or not. Each workshop followed the same procedure: (1) We started by reminding participants about the objective of the project and the methodology of FCM, and explained the method used to build the social maps. (2) We then presented the stakeholder map and asked whether this conformed with their perceptions. (3) We then modified the stakeholder map in real time to account for changes needed to represent a consensual vision among the group of mangrove socio-ecosystems in French Guiana (final variables are described in **Supplementary Table 1**). In addition to the new social maps, we collected interesting qualitative material based on the comments from stakeholders during the workshops.

The four modified stakeholders FCMs were then aggregated into a final community map. The final representation of FCMs was done using Mental Modeler software (available at: www.mentalmodeler.com; Gray et al., 2013).

Analysis of Fuzzy Cognitive Mapping

FCMs can be coded into adjacency matrices in the form $A(D) = [a_{ij}]$, where a_{ij} represents the strength of the effect of variable i on variable j. Variables identified in the map are listed both on the vertical axis (v_i) and on the horizontal axis (v_j). a_{ij} take a value between –1 and 1, with 0 meaning no connection.

From these matrices, different metrics can be calculated using graph theory, to help in FCM analysis (Özesmi and Özesmi, 2004; Bosma et al., 2017). Indeed, the adjacency matrix allows the use of algebra tools from graph theory to produce a series of indices characterizing map structure, that can then be compared using statistical analysis, including one-sided ANOVA. Examining the structure of the map allows us to determine how the respondents perceive the system. First, the number of variables $[N]$ and the number of connections $[C]$ are determined and used to calculate density $[D]$. When density is high, respondents perceive more relationships among variables and thus more options to change the system.

$$D = \frac{C}{N(N-1)}$$

The types of variables can also be examined to assess how they will interact. There are three types of variables: (1) transmitter variables [T] that designate forcing functions or endowments, variables that come as given and on which actors in the system have no power; (2) receiver variables [R] that refer to utility variables or ends, and that are the output of the system and; (3) ordinary variables [O] that represent the means by which the system can evolve. The ratio of the number of receiver variables (R) divided by the number of transmitter variables (T) measures its complexity (Özesmi and Özesmi, 2004). A high number of receiver variables illustrate a system with many outcomes or implications. On the other hand, a large number of transmitter variables indicates a system with top-down influences. In the end, complex maps will present larger complexity ratios (R/T), as they present more utility outcomes and less controlling forcing functions.

The contribution of the different variables is based on the calculation of their outdegree $[od\,(v_i)]$, indegree $[id\,(v_i)]$ and centrality $[td\,(v_i)]$ scores. Outdegree score is the row sum of absolute values of a variable in the adjacency matrix. It shows the variable's cumulative strength on other variables.

$$od\,(v_i) = \sum_{k-1}^{N} \overline{a_{ik}}$$

Indegree score is the column sum of absolute values of a variable in the adjacency matrix. It indicates how much a variable is influenced by other variables through the cumulative strength of variables entering the variable.

$$id(v_i) = \sum_{k-1}^{N} \overline{a_{ki}}$$

The overall contribution of a variable in a cognitive map can be understood by calculating its centrality (or total degree) $[td(v_i)]$ as the summation of its outdegree and indegree scores.

$$td(v_i) = od(v_i) + id(v_i)$$

The last metric that can be used to assess the structure of the map is the hierarchy index (h). When close to 1, a system is called hierarchical while close to 0 it is called democratic. Democratic systems are much more adaptable to local changes because of their high level of integration and dependence.

$$h = \frac{12}{(N-1)N(N+1)} \times \sum_i \left[\frac{od(v_i) - (\sum od(v_i))}{N} \right]^2$$

RESULTS

Individual Fuzzy Cognitive Maps

Table 2 presents the graph theory indices obtained for the 19 FCMs collected after the individual interviews. Interviewees identified a total of 257 variables to characterize French Guiana's mangrove systems, with a mean number ($\pm$ SD) of 13.53 ($\pm$ 6.26) variables per map and 15.26 ($\pm$ 8.39) connections. Note that 5 additional variables were mentioned without any connection to other variables; they are excluded from the following calculations.

A visual analysis of the indices suggests that economic actors identified fewer variables and connections. Moreover, in comparison to other categories, they identified a very low number of transmitters suggesting a high level of complexity with few controlling forcing functions. We ran a one-sided ANOVA on the indices that showed only one statistical difference amongst our stakeholder groups regarding the number of receivers (F = 33,156; df = 18; p = 0.049): economic actors and scientists perceive a below-average number of receivers showing a low number of "outcomes" from the system.

A first homogenization of variables allowed identifying several categories in which they could be grouped. Many variables have a very low number of records: 46 variables were mentioned only once and 20 only twice or thrice. 21 concepts were mentioned between four and eight times. Finally, five variables were mentioned by more than half of respondents, namely "Nurseries" (10 times), "Coastal protection" (11), "Biodiversity" (12), "Fishing" (13), and "Mangrove" (19).

Stakeholders Fuzzy Cognitive Maps

Before presentation of the results during the workshops we condensed the number of variables from 92 to 29. After aggregation of individual maps among stakeholders we obtained four stakeholder FCMs with a mean number ($\pm$ SD) of 21.5 ($\pm$ 4.7) variables and 34 ($\pm$ 13.5) connections.

A descriptive analysis of the graph indices of the different stakeholders FCMs (**Table 3**) shows that the FCM from economic actors presents the highest level of complexity, i.e., stakeholders perceive more outcomes from the system than options to intervene on it to make it change (given the number of identified transmitters). FCMs from conservationists present the highest density meaning that their perception of the system is the most interconnected with many links between variables. The study of the FCMs from scientists shows a more concentrated vision of the system with fewer variables (17) and a relatively high level of complexity. Finally, managers have the most extended perception of the system with many variables and a great majority of ordinary variables.

After the workshops, the mean number of variables across FCMs increased to 24.3 ($\pm$ 3.3) and the mean number of connections also increased to 40.3 ($\pm$ 11.7). Standard deviation decreased for all the graph indices, except for the number of receivers.

Community Fuzzy Cognitive Mapping

In this section, we present the community FCMs obtained from the aggregation of our four expert groups FCMs after the workshops. The aggregated community map is presented

TABLE 2 | Graph theory indices of the individual FCMs: mean and standard deviation by stakeholders group.

	Conservationists	Economic actors	Scientists	Managers	All
No. of maps	3	5	3	8	19
No. of variables (N)	15.67 ± 7.51	9.00 ± 2.65	13.33 ± 2.65	15.63 ± 6.95	13.53 ± 6.26
No. of transmitter variables (T)	2.33 ± 1.53	0.80 ± 0.84[a]	2.67 ± 0.84	2.25 ± 1.83[a]	1.95 ± 1.58
No. of receiver variables (R)*	7.33 ± 4.04	3.6 ± 1.95	3.67 ± 1.95	7.38 ± 2.67	5.79 ± 3.05
No. of ordinary variables (O)	6.00 ± 2.00	4.6 ± 2.88	7.00 ± 2.88	6.00 ± 3.63	5.79 ± 3.29
No. of connections (C)	17.67 ± 9.50	9.80 ± 3.96	16.00 ± 3.96	17.50 ± 9.43	15.26 ± 8.39
Complexity (R/T)	3.25 ± 0.66	4.17 ± 2.36[a]	2.22 ± 2.36	4.44 ± 2.82[a]	3.75 ± 2.35
Density (D)	0.08 ± 0.04	0.12 ± 0.03	0.09 ± 0.03	0.08 ± 0.03	0.09 ± 0.03

[a] *Include answers with no transmitter that forbid the calculation of complexity index.*
**Statistically significant differences in indices among stakeholder groups (p < 0.05).*

TABLE 3 | Graph theory indices of social maps obtained at different stage of the consultation process.

	Stakeholder FCMs before workshop				Community map		
	Conservationists	Economic actors	Scientists	Managers	#1—after interviews	#2—after condensation	#3—after workshop
No. of variables	21	20	17	28	92	29	30
No. of transmitter variables	5	3	3	3	11	2	2
No. of receiver variables	8	8	6	3	29	2	3
No. of ordinary variables	8	9	8	22	51	25	25
Number of connections	30	26	26	54	178	79	90
Connection per variable	1.43	1.3	1.52	1.93	1.93	2.72	3
Complexity (R/T)	1.6	2.7	2	1	2.63	1	1.5
Density	0.36	0.031	0.031	0.06	0.02	0.09	0.1
Hierarchy index	0.002	0.002	0.005	0.002	0.0001	0.004	0.02

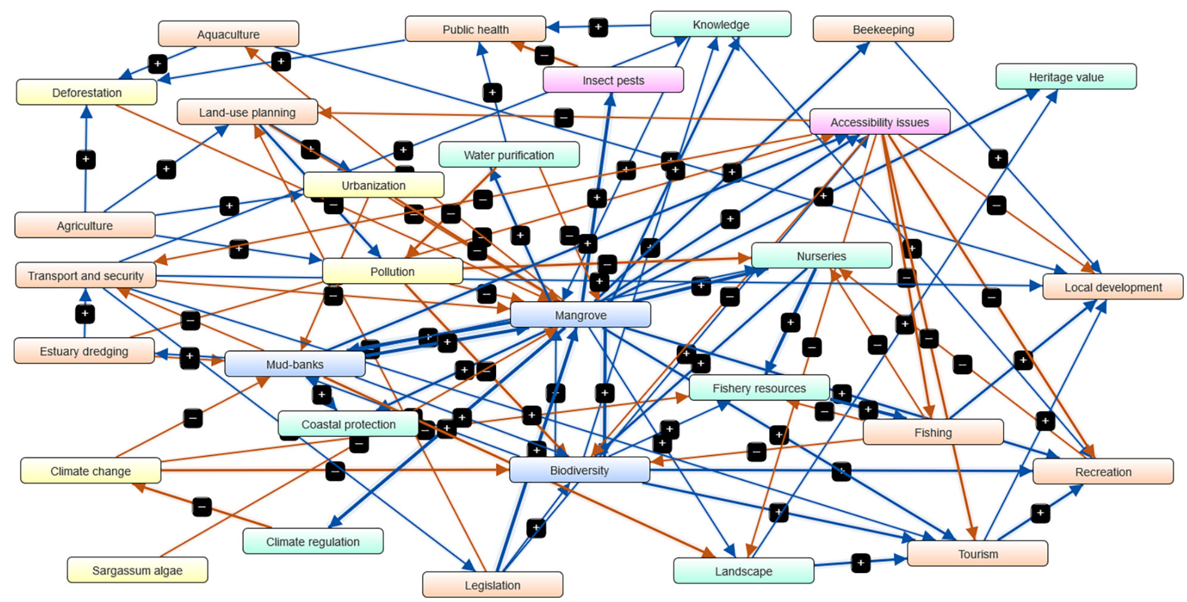

FIGURE 1 | Community map after workshop [Blue arrows (with a "+" sign) indicate a positive relationship, while orange (with a "-" sign) indicate a negative relationship. Thickness of the arrow reflects the strength of the relationship from | 1| (when reported by every group) to | 0.25| (when reported by only one group). Box colors are selected to assist the reading of the FCM: Blue boxes for biological compartments, green boxes for ecosystem services, pink boxes for disservices, orange boxes for anthropic activities and yellow boxes for pressure vectors].

in **Figure 1** and the corresponding graph theory indices are presented in **Table 3**.

The high number of ordinary variables indicates numerous interactions between system components. This outcome corroborates the important number of connections identified, suggesting perceptions of a strongly integrated and interdependent mangrove socio-ecosystem. That is, the system's dynamics result from mutual influences between the French Guiana society and the mangrove ecosystem. The low hierarchical index (close to 0) indicates a relatively "democratic system" with a high level of integration and dependence between the different components of the system. A system perceived as democratic is a sign that experts consider there are various ways to change it. Yet, the rather low density index reports a sparse map, reflecting that few management options are identified as capable of influencing the system's dynamics.

Table 4 ranks the community map variables by order of centrality. Variables with highest centrality scores are influential within the system: i.e., they can either be highly connected to other variables or display few connections with a high weight.

The most influential variables are associated to the biological compartments of the mangrove ecosystem, namely "Mangrove," "Mudflat/Mud-banks," and "Biodiversity," followed by "Nursery," "Accessibility issues" and to a lesser extent, "Recreational activities." Together, these elements come out as key influential dimensions of the socio-ecosystem's functioning. In accordance, the anthropic activities "Tourism" and "Fishing" come next. Tourism is most directly supported by biodiversity and as such, increases demand for recreational services, themselves relying on biodiversity, while fishing rests on mangrove supply of fishery resources, through its habitat support function. In turn, all three anthropic activity types are concerned by accessibility

TABLE 4 | Indices regarding variables of the community map.

Ranking	Variables	Indegree score	Outdegree score	Centrality score	Centrality change[a]	Ranking change[a,b]	Occurance	Occurance change[a]
1	Mangrove	4.75	10.5	15.25	–1.5	=	4	=
2	Biodiversity	4.25	2.5	6.75	0.75	=	4	=
3	Mudflat/Mud-banks	2	3.5	5.5	0.75	=	4	=
4	Accessibility issues	1.75	3	4.75	2.75	↑↑	4	↑
5	Nurseries	2.5	1.75	4.25	0.5	=	4	=
6	Tourism	2.5	1	3.5	1	=	4	=
7	Pollution	1.5	1.75	3.25	1	=	4	=
8	Recreational activities	3	0.25	3.25	0.5	=	4	↑
9	Fishery resources	1.75	1	2.75	1.25	↑	4	=
10	Fishing	1.5	1.25	2.75	–0.5	↓	4	=
11	Land-Use Planning	0.75	1.75	2.5	1	↑	4	=
12	Transport and security	0.75	1.25	2	0.5	↑	2	=
13	Legislation	0.25	1.75	2	0.25	=	4	=
14	Urbanization	1	1	2	2	→	3	=
15	Climate change	0.5	1.25	1.75	0.25	=	3	=
16	Water purification	0.75	1	1.75	0.25	=	3	=
17	Landscape	1	0.75	1.75	0.25	=	4	↑
18	Knowledge production	1	0.75	1.75	0.25	=	3	=
19	Local development	1.75	0	1.75	0	↓	3	=
20	Insect pests	1	0.5	1.5	0.5	↑	4	↑
21	Climate regulation	1	0.5	1.5	0	↓	4	=
22	Public health	1	0.25	1.25	0.25	↑	2	↑
23	Coastal protection	1.25	0	1.25	–0.25	↓	4	=
24	Agriculture	0	1	1	0.25	=	3	↑
25	Heritage values	1	0	1	0	↓	3	=
26	Deforestation	0.75	0.25	1	0	↓	1	=
27	Estuary dredging	0.25	0.75	1	0	↓	3	↑
28	Beekeeping	0.5	0.25	0.75	0.25	=	2	=
29	Aquaculture	0.25	0.5	0.75	0.25	=	1	=
30	Sargassum algae	0	0.25	0.25	0	=	1	=

[a]*Changes materialize a change in the community map following discussions during the workshops: "=" no change; "↑" increase; "↓" decrease; "→" new variable.*
[b]*Change for ranking is taking into account when superior to 2 ranks.*

issues arising from mud-banks and mangrove forest. These latest components can be detrimental to fishing vessels' access needs to the coastline and/or to the amenity value of beach landscapes sought by (certain) tourists. The influence of mangrove on recreational activities is more ambiguous, since these activities are supported by mangrove forest (for kayaking, bird watching, hunting, hinking, etc.), while they can also be negatively affected by the limited access forest and mud-banks impose on coastal areas (preventing watching the laying of turtles on the beach for instance). Pollution is the environmental impact factor displaying the highest centrality score. It is harmful to several benefits derived from the mangrove ecosystem, via its negative impacts on biodiversity and ecological habitats.

The centrality of the subset of variables presented above may owe to the fact that they are mentioned by four stakeholders' groups we surveyed who consider them important to the functioning of the system (**Table 4**). Indeed, looking at the very core of the representation shared by the expert community we interviewed (**Supplementary Figure 1**), it is clear that the mutual interdependence between mud-banks and mangrove forest is acknowledged by all experts, so as the role of the mangrove ecosystem in supporting biodiversity and the problems it causes in terms of accessibility. In relation to society's demands, only fishing activities stand out, through the channel mangroves foster "Nurseries" (support service) and "Fishery resources" (provisioning service).

Other variables emerge in **Supplementary Figure 1** as being mentioned by every category of stakeholders. Because of its relatively high outdegree score but low indegree score (id = 0.25 vs. od = 1.75), "Legislation" does not count as "very" central to the system's dynamics: it is perceived as a lever for mangroves conservation with direct and positive relationship with mangroves and biodiversity. On the other hand, climate regulation is a mangrove ecosystem service with a higher indegree score than outdegree score (id = 1 vs. od = 0.5). While expert groups are not unanimous and clear on the consequences of climate change on the mangrove socio-ecosystem, all agree on the fact that mangrove ecosystems play a role in climate regulation, in particular with respect to the carbon cycle. The variable "Insects issues" presents a similar behavior (id = 1 vs. od = 0.5) showing that all stakeholders have this issue in mind notably regarding the Yellowtail Moth (*Hylesia metabus*) that can provoke severe allergic reactions. Other ES are mentioned by all stakeholders but their connection with other variables of the system is less unanimous (namely "Coastal protection," "Landscape"). Finally, all stakeholders recognize the role of "Agriculture," "Pollution," and "Land-use planning" in the system but there is no consensus regarding the nature of this role. For example, in the case of "Agriculture": conservationists assumed an impact of agriculture on mangroves through territory planning, and an unclear effect of agriculture on mangroves through pollution carried by rainfall runoff; managers perceived a negative impact of agriculture via deforestation; scientists through land conversion; while the overall interaction was unclear for economic actors.

DISCUSSION AND CONCLUSION

Ecosystem Services Provided by French Guiana

All expert groups underline the role of mangroves in sustaining coastal fisheries. Indeed, the most unanimous relationship in the community FCM (**Figure 1**) is the inseparable link between mud-banks and mangroves, owing to "Nursery" functions necessary to maintain "Fishery resources." "Fishing" is thus perceived as the most emblematic activity depending on the mangrove ecosystem within the French Guiana society. In 2019, 2,820 tons of fish were landed in French Guiana, worth €5.4 million ($\approx$ US$ 6.1 million) (IFREMER, 2020). Acoupas (*Cynoscion* spp.) and Crucifix sea catfish (*Arius proops*) that represent the majority of catches of coastal fisheries (76% of total landed volume in 2019; IFREMER, 2020), spend part of their life cycle in mangrove (Rojas-Beltran, 1986; Rousseau et al., 2018). Also, mangrove fluctuation has shown to directly impact shrimp fisheries, that represented an important source of export for French Guiana (Diop et al., 2018). These numbers do not include illegal and subsistence fishing that could represent around 60% of total catches (Levrel, 2012). Mangrove conservation is thus critical for the sustainability of fishing practices, especially in the context of climate change that could lead to a collapse of both biomass of targeted species and fishing activities (Gomes et al., 2021). Meanwhile, fishing was reported by managers and scientists as an extraction activity that puts pressure on the natural environment. Fisheries can modify mangroves fish assemblages and impact their sustainability (Reis-Filho et al., 2019). However, empirically, it is currently unclear whether fishing has any impact on the quality of mangroves or the delivery of other services.

The importance of mangrove in "Climate regulation" is also recognized by all experts. Mangroves are very productive ecosystems that capture carbon from the atmosphere to develop. This carbon is then trapped and stored into the soil (Hamilton and Friess, 2018; Richards et al., 2020). In the case of French Guiana, the migration of mud-banks brings uncertainty to this role. During an accretion phase, the mud-bank is colonized by mangrove and starts to accumulate carbon (Marchand, 2017). When it enters an erosion phase, mangrove is destroyed and important quantities of organic matter are exported to coastal and offshore waters (Mongruel et al., 2018). Nevertheless, the total mangrove carbon stock at the scale of French Guiana is estimated to 23.06 $\pm$ 5.03 TgC (Walcker et al., 2018). Mangroves destruction would liberate this sequestrated carbon (Hamilton and Friess, 2018; Richards et al., 2020). As a signing party of the Paris Agreement on climate change, France has a duty in conserving this carbon stock.

There is a common agreement among our experts that mangroves have a potential role to play in the development of "Tourism" and "Recreational activities" (**Supplementary Figure 1**). In their review, Himes-Cornell et al. (2018) found that almost all the studies they selected provided economic values for recreation and tourism services by mangroves, confirming this role at the global level. However, we found no studies in French Guiana and experts during workshops underlined that eco-tourism in mangroves remains relatively marginal. This reflects the perception that mangrove and its biodiversity richness are identified as a potential for the development of the territory (WWF, 2017). Indeed, while only 21% of tourists come to French Guiana for leisure and 28% to discover the forest or the coast (CTG, 2016), there is a political will to increase those numbers relying on the development of eco-tourism (CTG, 2013). The final map provides the vision of a positive effect of mangroves on tourism and recreation that suggest that a development of these activities compatible with mangrove conservation is possible. However, the link between mangroves and the development of tourism and recreation is ambiguous (**Figure 1**). Firstly, because Recreation and Tourism encompass various practices that may be positively or negatively affected by mangroves. Certain activities (e.g., beach activities) can be negatively affected whether because the mangroves prevent access to areas of interest or because they modify the seascape. On the other hand, mangroves can also be attractive for eco-tourism and nature-related activities (e.g., faunistic observation). Secondly, because mangrove is closely related to mud-banks that are perceived as a constraint for the development of these activities. The migration of mud-banks can bring mangroves where they will be considered as a discomfort (e.g., on cities seafronts). In addition, there is currently a lack of infrastructure in French Guiana (e.g., there are only two marinas in French Guiana), developing tourism and recreation will imply new development projects.

Mangrove socio-ecosystems are also involved in the construction of coastal "Landscape" but the nature of this role is not obvious. If some experts consider that mangroves can have a positive impact on landscape, this is also negatively impacted by mud-banks and accessibility issues. Experts underline that urban seafronts are deserted when mangroves obstruct the seaview, while mangrove landscapes are appreciated in rural areas where they contribute to the identity of French Guiana. Aesthetic values of mangrove are understudied (Himes-Cornell et al., 2018). The question of the perception of the aesthetic value of a landscape is complex as it results from the link between the intrinsic characteristics of an object and its perception by an observer which is influenced by human nature, education and society (Tribot et al., 2018). According, to Tribot et al. (2018) there is a disconnection between the landscape aesthetic and the ecological value of ecosystems: mangroves are more likely to trigger negative perceptions than agricultural landscapes. The authors propose a virtuous loop in which, knowledge and experience on the functioning of ecosystems could increase mangroves' aesthetic value that would be more likely to be protected.

The last ES mentioned by every group of experts is "Coastal protection." The role of mangroves in providing coastal protection in French Guiana is closely associated to the dynamics of mud-banks as an alternation of "bank" and "inter-bank" phases. In bank areas, ocean energy is at first dissipated by the mud-banks, closer to the shore, the remaining energy is gently dissipated by mangroves (Anthony and Gratiot, 2012). Moreover, mangroves favor sedimentation (Furukawa et al., 1997) and enhance the resistance of the substrate during erosive inter-bank phases (Fiot and Gratiot, 2006). This process has been effective over the last 5,000 years, resulting in a net coastal progradation (Anthony et al., 2010). In addition, in inter-bank areas where higher energy waves result in mangrove destruction, mangrove trees still dissipate wave energy and thus contribute positively to coastal protection (Anthony and Gratiot, 2012). In French Guiana, coastal erosion and coastal flood are two issues well identified by the natural risk prevention plan (DEAL, 2015), which should advocate for mangrove conservation.

The experts also all pointed two disservices associated to the functioning of the mangrove ecosystem, namely "Accessibility issues" and "Insect Pests." "Accessibility issues" reflect the fact that some coastal facilities (e.g., slipways, fishing docks, touristic seaside infrastructures) may get directly obstructed by mangroves or see their value decrease because of the negative perception of mangrove in the vicinity. "Insect Pests" relate to the fact that mangroves shelter species that can impact public health, notably the moth *Hylesia metabus* that cause skin rashes (Jourdain et al., 2012). According to stakeholders, mangrove deforestation near residential areas occurred for sanitary reason in the recent past. This study shows that mangrove conservation can be exposed to a trade-off with mangrove destruction aimed at reducing such disservices. This trade-off is generally under-estimated in ES assessments of mangroves and comprehensive framework should be implemented (e.g., Knight et al., 2017).

The services mentioned by only some groups were listed (e.g., Knowledge production, Heritage value) and can be used in future discussions with local focus groups to further establish whether these warrant additional investigation, given their perceived importance by French Guianese residents and economic actors.

Threats on Mangroves in French Guiana

In order of importance given the respondent's perception (**Table 4**), the first perceived threat on mangroves in French Guiana is pollution, as mangroves are affected by wastewater near urban areas. Close to Cayenne, the impacts of pollution are visible on the microbial taxa of the mangrove (Fiard et al., 2022). Water pollution is mitigated by the purification ES from the mangrove that benefit society. Mangroves and their associated ecosystems act as natural sinks that trap all kind of anthropogenic pollutants (e.g., heavy metals, MacFarlane et al., 2007; Kulkarni et al., 2018; organic matter, Xiao et al., 2018; Adame et al., 2019). However, the sustainability of this purification role is questionable as high loads of pollutants can considerably modify mangrove ecosystems. Eutrophication in mangrove favor growth of shoots over roots and decrease their resilience (Lovelock et al., 2009), it also modifies the phytoplankton communities (Manna et al., 2010). There is a lack of information to conclude on the threat that chemicals represent for mangroves. Still, some studies has shown the impacts of chemicals on mangrove trees (particularly in early life stages; Lewis et al., 2011) and on the associated trophic network (Kulkarni et al., 2018). The question of water quality in French Guiana is the subject of a dedicated strategy established in 2017 that provides for public support dedicated to wastewater treatment (DEAL, 2017). If this plan is implemented primarily for sanitary reason, the improvement of water quality should benefit to mangrove ecosystems.

The second main threat raising concern among stakeholders is urbanization. Land-use planning, i.e., the extension of human infrastructures to face the increasing needs of society is positively related to this pressure on mangroves in the FCM. As population is concentrated on a 10–30 km wide strip along the littoral (Zouari, 2015), population growth may thus affect primarily this area and thus negatively affect the mangrove. Estuarine mangroves, which thrive on rivers, are considered as more subject to pressure from territorial development near urban areas. Indeed, they offer stable land that can support infrastructure, as is the case of a recent project of power plant construction near Cayenne, that has been criticized for its impact on mangroves and insufficient mitigation requirements (Autorité Environnementale, 2019). Coastal mangroves would be less affected, as they are under influence of mud-banks migration that make the coastline very unstable. The temptation to stabilize this dynamic system for infrastructure, urban and economic development may be associated with high risks (Jolivet, 2019). In neighboring Guyana, mangroves have been replaced, to make space for agriculture and aquaculture, and coastal protection is now provided by coastal dikes. This has considerably modified the sedimentary dynamics and the country is now exposed to erosion that can only be countered by expensive engineering solutions (Anthony and Gratiot, 2012). The integration of mangrove variability in land-use planning is thus necessary, in order to integrate its positive and negative effects in the best possible ways.

Agriculture is actually booming with an increase by nearly 38% of cultivated area between 2010 and 2019, in order to cater for the growing needs linked to the territory's increasing population. To maintain food self-sufficiency, 1,000 hectares should be turned into agricultural land every year (CEREMA, 2016). Total used agricultural land was 33 800 ha in 2019, with more than half concentrated in the west part of the country (DEAAF, 2020). However, there was no consensus on the impacts of such development among our stakeholder groups calling for more investigation. Farming in French Guiana combines traditional manual itinerant agriculture, breeding and mechanized agriculture for commercial purposes. Sorting out the incidence of these different practices and their expansion on mangroves' ecological states calls for further investigation as was pointed out by most of our expert groups.

The third threat on mangrove ecosystems identified by stakeholders is climate change. According to stakeholders, the impact of climate change does not directly affect mangroves but rather elements of the ecosystem, namely "Mud-banks," "Biodiversity," and "Fishery resources" (**Figure 1**). For most stakeholders (excepted one scientist), climate change was considered as a broad variable. However, it is difficult to summarize Climate change interactions with mangroves in a single variable as this corresponds to multiple factors of environmental changes (e.g., sea level rise, drought, change in salinity) that can have different—if not reverse—impacts on mangroves, depending on the context (Alongi, 2015; Lee et al., 2021). For example, in French Guiana, climate change intensifies the swell regime, which accelerates mud bank movements that should impact coastal mangroves. Further investigation is needed in this regard. As for perceived negative effects of climate change on biodiversity, these seemed to corroborate established knowledge on ocean warming and acidification risks, rather than being derived from local evidence.

Fourthly, deforestation has a very low importance in the community map, compared to its importance at the international scale (e.g., Richards and Friess, 2016). This may be explained by the absence of some extraction activities generally associated with mangroves. First, wood harvesting for construction, manufactured goods, firewood or coal is mostly absent in French Guiana. The few extractions of mangrove wood that serve handicraft productions or smoking processes are negligible in the global functioning of the socio-ecosystem as shown by their absence from the FCMs. Second, the expansion of aquaculture remains limited and takes place mainly in freshwater from coastal plains. A small amount of mangrove oyster farming currently takes place in Montsinéry (upstream from Cayenne). The mention of aquaculture in the FCM reflects the several scientific and governmental programs that aim at developing the sectors' potential. Given the threat that aquaculture has applied on mangroves worldwide (Naylor et al., 2000), this may require special attention.

Facing those threats, one main lever of action is identified by stakeholders, namely "Legislation." Legislation is supposed to be able to act directly on the enhancement of state of the ecosystem ("Mangrove" and "Biodiversity") or indirectly in reducing "Land-use planning" (**Figure 1**). A single positive feedback loop is identified from the ES of "Knowledge production," with a positive effect on mangrove. However, stakeholders perceive no variables with the opportunity to reduce the identified threats. This result is surprising as it reflects a command-and-control perception of conservation, rather than a vision developed following socio-ecosystems management principles (Ostrom, 2009). Indeed, solutions based on more flexible institutional arrangement can also increase the ecological outcomes of conservation, as well as its social and economic benefits (Scemama and Levrel, 2019; Bellanger et al., 2021). Nevertheless, imagining effective solutions for mangrove conservation needs to take good consideration of the multiple interactions between mangroves and societies, the associated positive and negative incentives for mangrove conservation, and the numerous sources of variability and uncertainty. In particular, there is a need for innovative solutions to better integrate the dynamism of the coastline in the future development of the territory.

Interest of Fuzzy Cognitive Mapping

FCMs were created by merging variables and connections raised by all experts in each group. Such merging can lead to overly complex system representations that potentially include artifacts associated with individual misconceptions or biases merged into group responses. The combination of individual interviews and workshops allowed to validate and increase the confidence in the credibility and relevance of the results (Teixeira et al., 2018). Moreover, the organization of the workshops allowed to increase consensus regarding the variables and the connections between them, as we can see with the reduction of standard deviation between the stakeholders FCMs. Finally, discussions during the workshops provided qualitative arguments that help understanding the FCMs configuration in light of the variability observed in the territory under consideration. Indeed, the final FCM summarizes the perception of the socio-ecosystem at the scale of the French Guiana. It does not explicitly represent the many sources of variability such as the differences between coastal and estuarine mangrove or the temporal variability associated to the migration of mud-banks, while the functioning of mangrove ecosystems and their ES is closely related to their biogeographic and geomorphological characteristics (Lee et al., 2014). However, the process of generating the map enabled identifying these differences.

Another risk with merging variables under broader concepts is to lose information (see e.g., the discussion on climate change or recreation and tourism). As a result, FCMs may fail to reflect some local variations. The final community map provides the perception of the expert groups consulted regarding such local variation and how they should adequately be captured, at the scale of the entire French Guiana. As a result, it provides a relevant overview of the perception of the functioning of the mangrove socio-ecosystem of French Guiana, and of the actual state of knowledge on this system. The organization of workshops with stakeholders allowed to collect qualitative material that can help to appreciate variability. It would be interesting to realize similar exercises on more restricted areas to focus on local issues.

The use of expert knowledge in ES assessment is considered one of the most popular ES assessment techniques today (Jacobs et al., 2015; Campagne and Roche, 2018). Indeed, it is particularly adapted to face the uncertainty-urgency dilemma that characterizes biodiversity conservation. As such, expert consultation results fit within a post-normal framework for ES assessment (Ainscough et al., 2018). In such a framework, expert knowledge is used to overcome uncertainty issues that can hinder conservation decision-making, to the benefit of a status quo detrimental to biodiversity and ES protection. Moreover, this is particularly interesting where the scientific evidence is not sufficient to support a comprehensive ES assessment as underlined by Mongruel et al. (2018). In such comprehensive assessments, economic analysis generally relies on the use of benefit transfer (e.g., Giry et al., 2017; Trégarot et al., 2021), using values commonly associated to mangroves in the literature and applying them to the studied territory. Our approach enables capturing the originality of the mangroves of French Guiana regarding the ES provided, in comparison to mangroves at the global scale, and shows that the use of benefit transfer without better knowledge of these key ES and disservices along with their variability would be hazardous at best. Based on our results, future research needs regarding mangrove ES in French Guiana, and their interactions with mangrove conservation policy can also be identified.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

AUTHOR CONTRIBUTIONS

PS was the main writer of the manuscript. ER, FB, and OT assisted in writing the manuscript. PS, ER, FB, and OT contributed to the research. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/ffgc.2021.769182/full#supplementary-material

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The Effectiveness of Financial Incentives for Addressing Mangrove Loss in Northern Vietnam

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This paper analyzes the effectiveness of existing financial incentives for mangrove conservation in Vietnam. Current conservation programs and projects have created financial incentives for mangrove protection, but the effectiveness of these incentives in addressing mangrove loss in northern Vietnam has been mixed. While financial incentives have contributed to a larger area of planted mangroves, their effectiveness is hampered by contradictory national policies, which encourage mangrove conservation on the one hand, and aquaculture expansion in mangrove areas on the other, thus making it difficult to address mangrove deforestation and degradation effectively. Mangrove conservation in Vietnam is challenged further by inequitable distribution of power and benefits, difficulties accessing information, weak law enforcement, lack of compliance, low payments for protecting mangroves, lack of full recognition of local rights, discontinued funding after policies and projects end, and lack of participation by local people in policy and project design and implementation. Conservation policies and projects should aim to protect existing mangrove forests, restore degraded mangroves and plant new ones to enhance mangrove area, quality and biodiversity. Sustainable mangrove conservation not only requires effective and sustainable financial incentives, but other enabling conditions such as addressing the conflict between mangrove conservation and aquaculture expansion, and grounding mangrove conservation projects by building on local knowledge and leadership. As these drivers are often motivated by national development goals and other sectoral development needs with ministries competing for budgets and influence, holistic land-use planning needs to be coupled with effective coordination and clarification of responsibilities between government agencies, and coordinated and consistent policies concerning these natural resources. Addressing these underlying governance issues is far more important for mangrove conservation and restoration than merely offering financial incentives as various national and international projects have attempted.

Keywords: mangrove, Vietnam, deforestation, financial, incentive

INTRODUCTION

The Warsaw Framework for REDD+ highlights the need to address the diversity of drivers of deforestation and forest degradation (UNFCCC, 2013). There has been increasing emphasis at both global and national levels on the need for more rigorous assessment of existing policies and initiatives (Duchelle et al., 2018; Pirard et al., 2019; Bos et al., 2020; Pham T. T. et al., 2020; Pham et al., 2021), as impact evaluations derived from past and present experiences can inform and improve new generations of policy and project interventions (Slootweg et al., 2001). Mangroves provide important environmental services for humankind, but are in decline globally and exposed to various drivers of deforestation and degradation (Giri et al., 2015; López-Angarita et al., 2016; Richards and Friess, 2016; Goldberg et al., 2020). To address these drivers, global initiatives like the Global Mangrove Alliance (Friess et al., 2020), REDD+ (Yee, 2010; Ammar et al., 2014; Aziz et al., 2016), payment for environmental services (PES) and national policy interventions (Wever et al., 2012; Locatelli et al., 2014; Friess and Thompson, 2016; Sommerville, 2016; Thompson et al., 2017) have been developed. However, these incentives have been impeded by unclear and contradictory regulatory frameworks across different levels of government, and failure to engage local people in policymaking and project design and implementation (Ahmadia et al., 2015; Friess et al., 2016). In addition, there is little independent monitoring and evaluation to assess the effectiveness of policies and projects.

To address these gaps, a large number of studies have been devoted to assessing the effectiveness of mangrove conservation projects, which often include a component on mangrove restoration policies and activities (Primavera and Esteban, 2008; Miteva et al., 2015; Kodikara et al., 2017; Ellison et al., 2020; Al Amin et al., 2021). To assess the effectiveness of these schemes, some authors have applied quasi-experimental techniques to compare what actually happened to what would have happened in the absence of any intervention (Ferraro, 2009; Miteva et al., 2015). Others have used a socio-economic assessment approach to unpack local perceptions on how effectively mangrove conservation policies and projects address mangrove loss on the ground while ensuring local livelihoods from aquaculture (Maliao and Polohan, 2008; Ahmadia et al., 2015; Mwangi et al., 2017; Santos et al., 2017), or have adopted remote sensing and spatial analysis, or *in situ* measurements and field observations (Seto and Fragkias, 2007; Li et al., 2013). Increasing attention is being paid to the need for more studies on the socio-political and institutional aspects of mangrove conservation efforts (Datta et al., 2012; Sahu et al., 2015; Apollonio et al., 2016; de Almeida et al., 2016; Dharmawan et al., 2016; Damastuti and de Groot, 2017; Triyanti et al., 2017; Singh et al., 2019; Turschwell et al., 2020; Mollick et al., 2021), as successful policy intervention depends on public trust and support from government agencies and projects (Harring, 2018), actors' ideology and interest and the political economy of drivers of deforestation and degradation (Brockhaus et al., 2021), political interaction by government with various interest groups (Oates and Portney, 2003), how benefits are shared (Pham et al., 2014a), local champions, the emergence of a crisis point, the involvement of decision makers and long-term financial and institutional support (Young et al., 2012), and how policies and projects align with local grassroots knowledge and perceptions (Dharmawan et al., 2016). Using case studies from Vietnam, this paper explores stakeholder perceptions on the effectiveness of existing mangrove conservation initiatives in the country.

According to the IMHEN and UNDP (2015), Vietnam, with its long coastline is vulnerable to climate change impacts. Although coastal forests cover just 3.5% of the total national forest area, they play a significant role in generating local livelihood incomes, mitigating the impacts of storm surges and coastal erosion, and supporting carbon sequestration and biodiversity conservation. Around 86% of the country's coastal 'protection forest' area constitutes mangroves (MARD, 2018). The Government of Vietnam recognizes the importance of mangroves in protecting coastal areas, and has issued several policies, including its Nationally Determined Contribution, the Vietnam Forestry Development Strategy 2021–2030 with vision to 2050, and Decree No. 119/2016/ND-CP, which place significant emphasis on protecting and expanding mangrove area in Vietnam (Pham and Nguyen, 2021). Through the Vietnam Forestry Development Strategy 2021–2030 with vision to 2050, the government aims to develop a sustainable mangrove forest management plan for the northern part of Vietnam and has called for more scientific analysis of lessons learned from past activities to provide input for this plan.

Specific investment programs responding to climate change have also been approved for implementation, including the Support Program to Respond to Climate Change (SP-RCC) and a project on the protection and development of coastal forests in response to climate change (Prime Minister of Vietnam, 2015). The Government of Vietnam sees mangrove conservation as a key policy for addressing climate change impacts and protecting coastal communities (Sam et al., 2005; Government of Vietnam, 2021). Sustainable mangrove conservation for climate change also marks a key future policy and investment priority, as per the approval of the National Climate Change Adaptation Plan for 2021–2030 (Prime Minister of Vietnam, 2020), and Vietnam Forestry Development Strategy for 2021–2030 (Prime Minister of Vietnam, 2021), which aims to ensure that forest area accounts for 42–43% of the country's total land area. Sustainable mangrove conservation is also an important measure in Vietnam's Nationally Determined Contribution (NDC) (UNFCCC, 2020), with a follow-up national program currently in preparation on investment for 2021–2025 in the protection of coastal forests and improving local livelihoods for coastal communities using mangroves. Sustainable mangrove conservation, as stated in Vietnamese policies, refers to mangrove conservation policies and projects that not only expand the area of mangrove forests, but also enhance their quality and capacity to support coastal communities. However, failing to connect future policies and programs with lessons learned from previous experiences could result in ineffective, inefficient and inequitable outcomes.

Various studies have attempted to investigate the effectiveness of policies and measures to address the drivers of deforestation and degradation in Vietnam. However, these have often assessed national policy initiatives while overlooking sub-national programs and community-led projects (Hawkins et al., 2010; Jhaveri et al., 2018; Kissinger, 2020; Pham T. T. et al., 2020). Using comparative case studies in Thanh Hoa, Quang Ninh and Thai Binh provinces, this paper analyzes the effectiveness of existing financial and policy incentives, at both national and sub-national levels to address the drivers of mangrove deforestation and degradation. Our aim is to provide lessons learned that could be useful for future mangrove conservation programs.

MATERIALS AND METHODS

Study Sites

The study was conducted in six villages in the provinces of Thanh Hoa, Thai Binh and Quang Ninh in northern Vietnam (**Figure 1**). Vietnam's administrative structure is subdivided into three levels: provincial, district and commune. Communes consist of a number of villages. The northern part of Vietnam was selected as it is strongly affected by climate change. The three provinces have been placed under national mangrove protection projects due to their shrinking area of mangrove forests, which have been in decline since the 1960s (Bui et al., 2014; Pham H. T. et al., 2020). However, few studies to date have examined the effectiveness of these projects or provided lessons learned for future mangrove conservation policies. Drivers of mangrove deforestation and degradation are analyzed in detail in Section "Drivers of Mangrove Deforestation and Degradation."

To address this knowledge gap, we selected these study sites to represent different aspects of mangrove conservation in Vietnam, including government management regimes, forest ownership, local income sources, accessibility to mangrove resources and land, and prior experience of mangrove management (**Table 1**).

Local people in these six villages rely primarily on aquaculture (Pham et al., 2019a). Livelihood sources include incomes derived from mangroves such as payments from programs and projects to patrol mangrove forests, seafood harvesting, honey farming, hunting birds, collecting timber and firewood from mangroves, ecotourism, and aquaculture. People from all six villages also have off-farm incomes from migrating to work in other cities or provinces (Kelly and Adger, 2000; Pham and Yoshino, 2011).

Methodology

The study employed a variety of methodologies. We first reviewed legislative and policy environments, and past and present mangrove conservation programs. We then conducted interviews with 34 key informants in the three study provinces (**Table 2**). Key informants included: ten government officials; six mass organization representatives, including women's unions and farmers' associations; two civil society organizations; and the six village heads. These stakeholders were selected because they were directly involved in the design and implementation of policies and projects. These interviews aimed at understanding stakeholders' perceptions of the drivers of deforestation and degradation in their areas, the effectiveness of national and foreign mangrove conservation policies and projects in addressing these drivers, and their recommendations for future policies and projects.

We also conducted 24 focus group discussions (FGDs) with a total of 240 participants across the six villages, and household interviews with 604 households in total (**Table 3**). We held four FGDs in each village: one with young women between 16–25 years old; one with women over 25 years old; one with young men between 16–25 years old, and one with men over 25 years old. Participants in these FGDs, averaging 10 household representatives per group, and surveyed households were selected randomly from official lists of villagers provided by commune people's committees. FGDs and household surveys covered similar discussion topics, including drivers of deforestation in the study sites, policies and projects implemented to address these drivers, and their effectiveness in achieving their intended objectives. To ensure people were active in discussions and no one taking part in the FGDs felt excluded or was inattentive, we used participatory visual communications tool such as pictures, cards and drawings. FGDs were only organized with villagers without village heads in attendance to ensure villagers felt free to express their thoughts. Survey instruments are fully presented in the **Supplementary Material**.

The FGDs and interviews were designed and conducted adhering to the ethical standards established within the scientific community. All questions used in FGDs, household and key informant interviews were reviewed by the local governments as part of the process to obtain research permission to work in the studied sites. The research objectives, data collection process, confidentiality by guaranteeing the anonymity of all villagers and key informant respondents were explained to the participants, and their consent was explicitly asked for.

RESULTS

Drivers of Mangrove Deforestation and Degradation

Participants in FGDs and household surveys in all six villages pointed out complex drivers of mangrove deforestation and degradation, ranging from natural drivers like typhoons and storms to complex socio-economic drivers rooted in national and provincial socio-economic development policies (see **Figure 2**). Among these, participants in FGDs and household surveys ranked the six main drivers as follows: natural disasters; mangrove conversion for aquaculture due to government policies promoting economic development and local needs to improve livelihoods; poverty; water pollution due to chemical waste from thermal plants and plastic waste from residents; migration; and limited awareness of local people on forest protection.

Natural Disasters

Ninety-two FGD participants mentioned natural disasters, such as typhoons, landslides and storms as being major drivers of mangrove loss. Participants in Thai Binh said they have observed increasing numbers of storms and floods since the 1990s. FGD participants in Thai Binh and Quang Ninh provinces also said that where two major storms a year used to be the norm, four

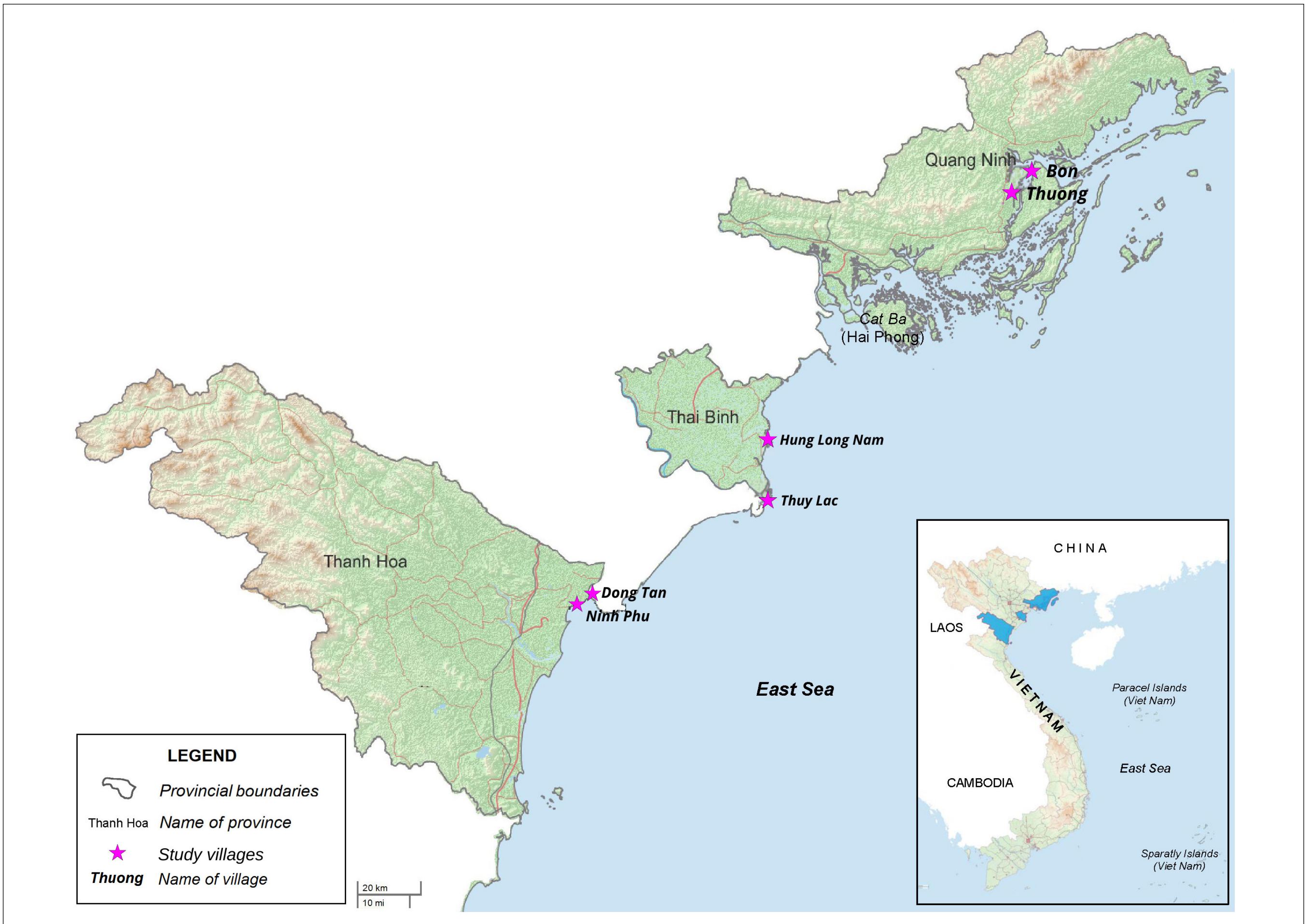

FIGURE 1 | Study location map.

or five a year has become commonplace in recent years. These FGD participants also pointed out that natural disasters have damaged large areas of mangrove forests. This was confirmed by many sources including government reports (VOV online, 2013), donor reports (Nguyen, 2003; IFRC, 2012; Vietnam Red Cross et al., 2013; MARD, 2017), media articles (VNA, 2016; Nguyen, 2020) and scientific reports (Veettil et al., 2019). They also noted that storm impacts on housing were less severe in areas with mangroves than without. Most of the households surveyed (60%) and key informants interviewed (48%) also referred to natural disasters as drivers of mangrove loss (**Figure 2**).

Mangrove Conversion for Aquaculture and Development Programs

More than 90% of households surveyed in the six villages cited aquaculture production as their primary source of income and admitted they only stopped being involved in mangrove clearing 5 years earlier. Conversion of mangroves for aquaculture was cited as the main driver of mangrove loss by 76% of key informants interviewed, 50% of households surveyed and 63% of FGD participants (**Figure 2**). Twenty-three out of 34 key informants interviewed, 100% of FGD participants and 30% of surveyed households said aquaculture development has always been a priority of the provinces, and provincial people's committees had issued many policies encouraging local people to clear mangroves for aquaculture development between the 1970s and early 2000s. As one FGD participant explained, "*I'm not originally from this province. I migrated from my hometown in Hai Duong province to Thai Binh in 1997 as part of the government migration program to promote economic growth. When we first arrived here, there was a large area of mangroves. Government officers encouraged us to clear as many mangroves as possible to develop and expand aquaculture production. We even received certificates showing how much mangrove we were able to convert for aquaculture production.*" One interviewed government official also said, "*Aquaculture can generate high incomes for local people, so in the past when the province was still poor and we had little economic development, previous leaders issued numerous policies on converting mangroves for aquaculture production.*"

The need to protect mangrove forests has only been emphasized in provincial policies over the last 10 years, but drivers like infrastructure development and conversion to aquaculture by government agencies and large-scale companies to meet provincial economic development goals still persist.

TABLE 1 | Background information about the study villages.

Village	Ethnicity	Main income sources	Mangrove area (ha)*	Formal forest managers	Who patrols forests?			Who has exclusion and prosecution rights?		
					Commune people's committee	Local community and people	Commune border police	Commune people's committee	Local community and people	Commune border police
Dong Tan	Muong, Dao, Kinh	Aquaculture and livestock	220	District people's committee**	x	x	x	x		x
Ninh Phu	Kinh	Aquaculture and remittance	70	District people's committee	x	x	x	x		x
Hung Long Nam	Kinh	Aquaculture, agriculture and remittance	170	District people's committee	x	x		x		
Thuy Lac	Kinh	Aquaculture, ecotourism, factory work	150	District people's committee	x	x		x		
Thuong	Dao and Kinh	Aquaculture and factory work	600	Commune people's committee	x	x		x		
Bon	Dao and Kinh	Aquaculture and ecotourism	677	Local communities		x		x		

Source: Data compiled by the authors in 2018.
**Mangrove forests in these six villages have all been planted since the early 1990s.*
***Vietnam's local administration is organized at three levels, provincial, district and commune. Provincial and district People's Committees can establish forests that are not of national importance and allocate use rights for all types of forest. Provincial People's Committees can designate rights to economic organizations or other organizations, while District People's Committees can designate rights to communes, households, individuals, and communities.*

Similarly, road construction and dike management have been positioned as priorities for developing provincial economies, and provincial leaders saw them as unavoidable compromises.

Poverty

Figure 2 shows 29% of FGD participants mentioning poverty as the main driver of mangrove deforestation, while only 9% of key informant interviewees and 10% of households surveyed cited poverty as a key driver of mangrove destruction. Eight out of ten government officials and all FGD participants, cited poverty as having been another driver of deforestation in the past, saying due to a lack of income and employment opportunities, local villagers had needed to clear mangroves for firewood to sell, expand grazing areas to increase income from livestock, and for ease of access to harvest seafood. Household interviews provided similar results, though they also showed clearing mangroves for firewood had stopped 10 years earlier, and nowadays people have better access to electricity and no longer need firewood for cooking and selling.

TABLE 2 | Numbers of key informant interviewees in the study sites.

Province	Total Number	Number of men	Number of women
Thai Binh	17	16	1
Thanh Hoa	11	7	3
Quang Ninh	6	4	2
Total	34	27	6

Water Pollution

While mangrove conversion for aquaculture, natural disasters, migration and poverty were seen as common drivers across the six villages, water pollution was specific to Thuong and Bon villages in Quang Ninh province and Hung Long Nam village in Thai Binh province. Government officials and all FGD participants in Quang Ninh and Thai Binh pointed out that

TABLE 3 | FGD and household interview sample sizes in the study sites.

#	Province	Study village	Mangrove area (ha)	No. of FGD participants	No. of households interviewed	% of female respondents	% as female heads of household
1	Thanh Hoa	Dong Tan	220	38	100	73	22
		Ninh Phu	70	42	100	40	18
2	Thai Binh	Hung Long Nam	170	41	101	86	11
		Thuy Lac	150	39	101	61	14
3	Quang Ninh	Thuong	600	39	103	89	24
		Bon	677	41	99	57	9

Sources: Central Population and Housing Census Steering Committee (2010); Dong Rui Commune People's Committee (2017); Nam Phu Commune People's Committee (2017); Da Loc Commune People's Committee (2018); Dong Long Commune People's Committee (2018); VNFOREST (2018); and Thanh Hoa Provincial Department of Agriculture and Rural Development, 2017.

the Thai Binh thermal power plant, Ba Che Paper Mills and Mong Duong thermal power plant have caused serious water pollution, leading to the destruction of mangroves. According to one FGD participant in Thai Binh, "*Thai Binh Thermal Power Plant had no wastewater treatment, so their discharge polluted water surrounding the village and ran through the mangrove area causing both new seedlings and existing mangroves to die.*" Informants cited plastic waste as another major cause of water pollution and mangrove degradation, with one FGD participant in Quang Ninh stating that, "*Residents throw waste and plastics directly into the sea. Small fishing platforms also dump plastic bags and waste into the sea. In fact, if you walk through the mangrove area in the afternoon or early morning you can see nothing but plastic. We do think this affects the health of mangroves.*"

Migration

While migration was seen as the main driver of mangrove deforestation by 21% of FGD participants, only 1% of households interviewed and 3% of key informants interviewed perceived migration as a key problem driving mangrove loss (**Figure 2**). All six key informant interviewees in Quang Ninh province said increasing numbers of migrants from other provinces (Ha Nam and Nam Dinh) had moved to Bon and Thuong villages to invest in aquaculture production. Most households interviewed in Dong Tan (67%), Thuong (58%) and Bon (88%), and some in Ninh Phu (20%), Hung Nam Long (14%) and Thuy Lac (17%) were migrants. These households said the government had encouraged them to migrate to these provinces to clear mangroves for the expansion of economic development zones. They said these migrants are often wealthy families with the financial resources necessary to bid for large areas for aquaculture production. They can also rent large areas of land from local people to open aquaculture farms and hire local people to clear mangrove forests. There are big gaps in terms of income between rich and poor households, and household incomes vary both within and across the six study sites. In Hung Long Nam village, FGD participants said rich households could earn VND 738 million a year on average, while poor households could only earn up to VND 30 million a year. Incomes of villagers in Hung Long Nam were twice those of villagers in Dong Tan. More than 30% of interviewed female heads of households and all female FGD participants in Hung Nam Long village said that since aquaculture ponds surrounded by mangroves are being privatized by rich migrants, they now have less access and fewer earnings derived from fisheries and can only harvest in small public mangrove areas.

Limited Awareness Regarding Forest Protection

Limited awareness of local people on forest protection and forestry policies are cited by 32% of household surveyed, 18% of key informants interviewed and 42% of FGDs meeting (**Figure 2**). Government officials interviewed in all three provinces claimed people had limited understanding of the roles of mangroves and government policies on mangrove protection. One government official interviewed in Quang Ninh stated that "*Local people don't know about the importance of mangroves, and lack awareness of government policies on mangrove conservation.*" Government officials in all three provinces said in cases of violations where local people had been arrested or fined for mangrove destruction, they claimed had they were unaware of government policies or who mangrove forest owners were. Despite these government officials pointing to ignorance over mangrove conservation rules and which government agencies (provincial, district or commune) owned mangroves, most participants in FGDs in the six villages, as well as 83% of households interviewed in Thuy Lac village and 76% of those in Bon village did know which government agencies were mangrove owners and what mangrove conservation policies were being implemented. One mangrove conservation participant in Thuy Lac village said, "*We've participated in many government-run training sessions on mangrove protection policies, so we know what they are. Moreover, we listen to the radio and watch television, and we understand we need to protect mangroves for our future benefits.*" There were no differences of opinion between men and women or between different ethnicities in regard to drivers of mangrove deforestation. Male and female FGD participants across all sites shared similar views on drivers of mangrove deforestation. These are outlined in **Figure 2**.

The Effectiveness of Financial Incentives in Addressing Drivers of Mangrove Deforestation and Degradation

All six study villages have access to and have prior experience of mangrove management from state and foreign mangrove conservation programs (**Table 4**). Mangrove planting in the three provinces has mostly been carried out through international and domestic reforestation programs. Mangrove forests in the provinces are either managed by district Forest Protection and Development Departments' Project Management Units (PMUs), which sign annual mangrove protection contracts with communes or other organizations (in Thanh Hoa) or managed by Provincial People's Committees (PPCs) and their representative, the Department of Agriculture and Rural Development (DARD) (in Quang Ninh and Thai Binh). DARD signs annual contracts with communes to protect the mangroves within their jurisdictions, providing funds from the state budget to cover costs. Communes also carry out mangrove rehabilitation funded by provincial authorities or with support from donors.

When asked "What are the changes in terms of mangrove area in your village?," 98% of households surveyed across the six study villages agreed that the area of mangroves had grown in their regions over the last 5 years as a result of programs run by international donors such as The Japan International Cooperation Agency (JICA) and the European Union (EU), and as a result of increased awareness of the roles mangroves play.

All participants in focus group discussions held in the six study villages said these state and foreign programs have provided financial support for local people to plant and patrol mangrove forests, training on the development of alternative livelihood options, and raised awareness on the roles and importance of mangroves. While these villagers felt this financial support has had positive outcomes, no official government data on its effectiveness was available. **Figure 3**, which was generated from remote sensing data coupled with ground truthing shows no

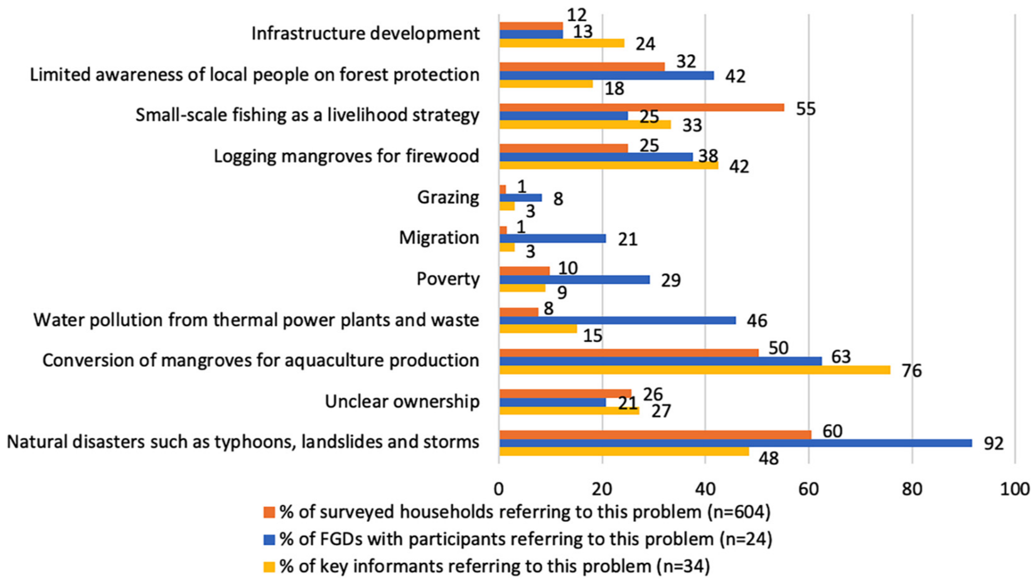

FIGURE 2 | Stakeholder perceptions on the drivers of mangrove deforestation in the six study villages.

TABLE 4 | Policies and financial incentives for addressing drivers of mangrove deforestation and degradation.

Policies/incentives	Government programs	International programs
Payments under annual forest protection contracts signed with local people.	VND 50,000–100,000 per hectare per year before 2010 (661 programs) and VND 3,000,000 per hectare for 6 years since 2017	Only pay for planting, but not for protection activities
Support for protecting special use forests	VND 30,000,000 per hectare	Not applicable
Support for mangrove conservation	VND 1,100,000 per hectare per year	Not applicable
Support for the establishment and operation of nurseries for seedling production	VND 75,000,000 per nursery	Not applicable
Payments for mangrove planting labor	VND 120,000–150,000 per day	VND 20,000–150,000 per day
Organizing waste collection activities in mangrove and coastal areas	No payment, community social responsibility	Some projects mobilize communities, but some pay VND 50,000 per day for local people to collect waste
Providing free seedlings	Yes	Yes
Setting up financial mechanisms to cover mangrove planting and protection		Local villagers contribute VND 2,000 each time they enter mangroves to collect fish and crabs, and fines are collected in Dong Rui commune (Quang Ninh) from people penalized for violating laws and for illegal destruction of local community mangroves. In Thai Binh province, people who report violations and violators will receive rewards of VND 200,000 for each report made.
Non-monetary incentives	Provision of training on government policies relating to mangrove conservation	Provision of free fertilizers, livestock and capacity building/training for alternative income options

Sources: Information compiled by the authors from reviews of government policies and project documents, and from FGD and household survey results.

increase, and in reality the areas of both natural and planted mangroves remaining relatively stable between 2016 and 2019 in the five communes where our study villages are located.

Stakeholders participating in FGDs and household surveys highlighted a number of benefits resulting from the mangrove protection policies now in place. They noted, for example, how these policies have helped strengthen law enforcement and raised local awareness on the roles forests play and the importance of their conservation, and have limited conversion of mangroves for other economic activities. According to FGD participants in Hung Long Nam village, for instance, before 2017, no villagers took part in mangrove forest patrols. However, since 2017, villagers have been receiving VND 87 million from government and foreign projects to cover time spent patrolling mangroves. The village has decided to use 15–20% of these funds to cover patrol costs for villagers to monitor forests, while the remainder is used for paying villagers to replant mangroves. Each participating villager earns USD 20–50 a year for their forest protection efforts. According to most villagers surveyed in Hung Long Nam, this additional income has been a major reason for people becoming

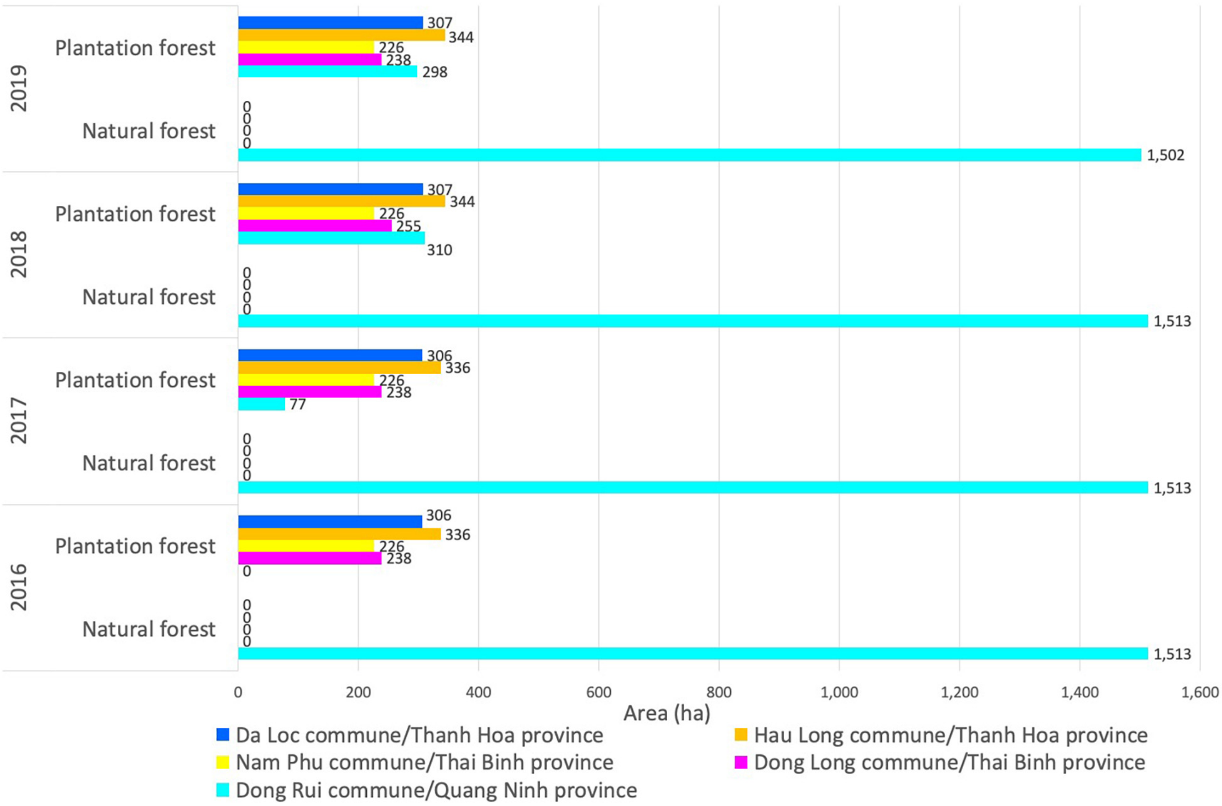

FIGURE 3 | Changes in forest area in the study communes. Source: Institute for Forest Ecology and Environment (2020).

involved in mangrove conservation. FGD participants in all six villages said more frequent forest patrolling had helped reduce illegal mangrove logging in their village areas.

Government and international projects have also provided capacity building and free seedlings for mangrove reforestation in the study sites. While government projects only provide financial support, provincial authorities and FGD participants across the six villages said non-state projects offered in-kind payments, such as livestock and training for local people on improving livelihood options. For example, training was provided on sustainable seafood production and techniques for rearing livestock more effectively. According to most surveyed households, such training was actually more important and useful to them than receiving monetary benefits. One household respondent said, "*Thanks to international projects, we've had opportunities to get training from experts on rearing livestock or seeking new income opportunities, and this is more useful for us comparing with in-cash payment.*" Interviews conducted with women's unions, youth unions or farmers' associations also revealed that their members (both men and women) had actively supported planting new mangroves. According to the women's union in Dong Rui commune, they had been nominated to lead a mangrove planting project funded by the Commune People's Committee.

FGD also showed that while local people derive more than 70% of their income from aquaculture and they had cleared mangrove in the past for this purpose, participants also pointed out the downsides of aquaculture. More than 80% of male and female participants in the FGD in Hung Long Nam said shrimp farming had caused serious soil erosion in their region. Male and female participants in the FGD in Thuy Lac both said only a small number of wealthy households could open, expand and benefit from their own private aquaculture ponds, and because mangrove forests are being privatized, poor households are restricted to accessing only public mangrove forests. Men and women participating in the FGD in Dong Tan village both said that due to a lack of access to aquaculture ponds, conflicts had intensified between wealthy migrant families and local people. As a result, many FGD participants were not in favor of clearing mangroves for aquaculture production.

Pitfalls of Incentives for Addressing Drivers of Mangrove Deforestation and Degradation

Although incentives have to some extent motivated local people to protect mangroves, our findings also reveal several pitfalls hampering effective, efficient and equitable mangrove protection and development.

First, most government officials interviewed claimed that overlapping mandates and responsibilities over mangrove management between government agencies and between different levels of government pose challenges to implementing mangrove conservation policies and projects. **Table 1** shows multiple government agencies involved in managing mangroves, including district people's committees, commune people's committees, and border police. However, these officials admitted to an unclear division of tasks and responsibilities between these actors. This leads to weak law enforcement on the ground, and more than 40% of households interviewed in the six study villages said they were unsure who manages their mangroves and what government agency they need to report violations to. Villagers interviewed said they lack ownership over current mangrove protection projects and programs and therefore feel no strong commitment to such efforts. These stakeholders claimed that mangrove forests are government property and therefore land-use rights belong to the government. More than 60% of households interviewed across the six villages said their commitment to protecting mangrove forests was low, as they were just providing labor for projects. One villager in Quang Ninh explained this saying, "*We're hired labor for someone else, so we just do what we're paid to do. It would be different if this forest was our property. If it was, we'd protect it and manage it full time. At the moment, as it's not our land, if they pay us to patrol it one or 2 days a month, then that's what we'll do.*" Even in study villages in Quang Ninh province, where the government has allocated mangroves to villages for protection, and communities have their own rules requiring local people to contribute VND 5,000 to village funds each time they visit public mangroves to fish or collect seafood, the village heads interviewed claimed such initiatives had been introduced by foreign projects without building on local ownership. As one interviewed village head explained, "*This was introduced to us by a foreign program. Of course, while their project is still active in the region we'll follow this model as get financial support to do so. But I'm not sure if we'll continue this model once the project ends as it wasn't developed by us.*" They also said that although local government had allocated mangroves to communities, they do not have full land-use rights. One villager elaborated on this saying, "*We're allocated mangroves but not the rights to use or benefit from them. In fact, rights only come with the responsibility to protect them more, with no associated benefits.*" All household survey participants in Thanh Hoa and Thai Binh provinces also said that when government agencies and officials selected households to join mangrove forest conservation programs, they would often choose their relatives or those with power, such as village heads or community party secretaries. As a result, many poor households cannot participate in or benefit from such policies and projects. FGDs revealed that international projects have been ineffective in disseminating information to local people. Respondents in most villages said only elite groups (village police) were informed about project activities and had opportunities to access related payments. As one villager in Thai Binh said, *"Our village head receives information, which he makes sure his family members and relatives hear first so they can register themselves in government programs. We don't know about these programs to register."*

Second, FGD and household survey results show that most incentives provided by state and non-state institutions are aimed at improving seedlings and offering financial incentives for locals to plant mangrove forests (**Table 4**). However, most FGD stakeholders and interviewed government agency staff said neither state nor non-state programs provided seedlings that suit local conditions. As a result, seedling survival rates had been low, and the quality of planted mangroves failed meet local expectations. As one villager explained, "*Government agencies and international projects provide us with seedlings to plant mangroves, but they're not the traditional species we've seen before from natural forests used by our ancestors. They don't seem to grow very well in our village.*" Further, none of these initiatives have aimed to address drivers of mangrove loss and degradation, such as government policies to convert mangroves for infrastructure development or aquaculture production. Most (70%) of the government officials interviewed claimed it is easier for the government to improve technical issues rather than address political needs rooted in the requirement to secure better economic returns from the land. One district government agency interviewee said, "*We're told to protect mangrove forests but we're also tasked with showing increased annual GDP and boosting the economic returns from the aquaculture sector by expanding aquaculture production. As economic development is our province's priority, our provincial people's committee gave more favorable support to expand aquaculture and we, at the district level, have no power to change this and have to follow decisions made at a higher level.*" One provincial government interviewee said, "*It is up to national and provincial authorities to expand our land to the sea and develop new ports and now coastal economic development zones. Consequently, we have to clear mangrove forests for these development priorities.*"

Third, current incentives are mainly aimed at planting new mangroves; there are no strong incentives to conserve existing mangroves (**Table 4**). For example, between 2014 and 2020, the national Support Program to Respond to Climate Change (SP-RCC) allocated VND 412.7 million for forest protection activities but allocated 7.7 times that amount (VND 3,195 million) for planting new forests. Most government agency staff interviewed and FGD results in the three provinces revealed that actors receive higher financial incentives for replanting than for protecting existing mangroves. For example, as discussed earlier, each household can only earn VND 1 million a year for patrolling forests, whereas FGD participants in all six villages said they can already earn from VND 1,050 million to 1,5 million in 7 days replanting mangroves. In interviews, commune government officials also said new plantation programs would pay costs for planting and at least 1–2 years of maintenance. Consequently, local people taking part in FGDs said they prefer engaging in projects establishing new mangroves to those protecting existing mangrove forests.

Fourth, according to FGD participants and government agency stakeholders, the costs of mangrove conservation in the study sites are significant; the mangroves are affected by coastal erosion, regular tidal inundations and sandy soils. With such conditions, all interviewed government officials said costs for planting mangroves vary between VND 90 million and VND 500 million per hectare. Yet, government incentives only cover around a third of these costs. Financial incentives offered by state and non-state programs are also unable to compete with the amounts earned from aquaculture production. For example, according to FGD participants in Quang Ninh and Thai Binh, on average, selling seafood can provide women with daily incomes of VND 50,000 to VND 1 million, while men, who have longer work hours can earn at least 30% more than women. Meanwhile, the daily payment received for either planting or patrolling mangrove sites is just VND 20,000–150,000. Men and women interviewees and FGD participants both felt the payments from national and foreign projects were too low. One male FGD participant in Thanh Hoa said, "*They only pay us VND 20,000 a day to plant mangroves while I can earn up to VND 500,000 a day gathering seafood. The commune people's committee told us we plant mangroves as part of our social responsibility, and the small payments are aimed at helping us buy food and drinks rather than compensating us for our time and labor. However, we need to earn income first before we can practice our responsibility.*" In addition, even though some projects cover the costs of planting mangroves, they lack follow-up incentives or support for conservation or protection of planted mangroves. This might help explain why only 30% of households interviewed across the six villages are currently participating in any mangrove conservation programs. FGD results in all six villages also showed that young people are less engaged in mangrove conservation projects because they migrate to work in the cities, feel a lack of connection to mangroves and therefore are not interested in their protection. Government agency staff in all three provinces also said that as soon as foreign projects ended and payments for local people to patrol mangroves stopped, local people would stop protecting and patrolling mangroves.

Fifth, both male and female FGD participants in Thanh Hoa and Quang Ninh provinces claimed that there is a lack of involvement of local people in the design and implementation of mangrove conservation policies and projects, and therefore their activities do not fully address local needs. In these two provinces, 40% of households interviewed claimed that government agencies and projects had not consulted local communities about the management and conservation of mangroves before pursuing activities. According to most participants taking part in the six village FGDs, incentive designs also overlook gender aspects. Results from FGDs in the six villages show that women engage directly in seafood collection in mangrove areas, while men often go offshore fishing and migrate to cities for work. However, government and foreign projects often prefer men to take part in patrol teams, and it is household heads, who more often than not are men, who are invited to government and project meetings. According to most female heads of households interviewed, this arrangement excludes women from participating in and benefiting from government and foreign projects. As one female head of household explained, "*Most of the time it's us women who spot violations because we collect seafood in the mangrove area and often call the village head and men to arrest violators. Also, when we see people from other provinces clearing mangroves, we ask them not to. But when it comes to government and project meetings on forest protection and payments for patrolling mangroves, we're not included which I think's unfair.*"

Finally, according to most government stakeholders and local people interviewed, state and non-state initiatives take a "carrot and stick" approach. Incentives offered to local people include financial incentives (e.g., payments for planting and protecting mangroves), while disincentives include bans on mangrove destruction activities such as clearing forests for aquaculture and poaching. Fines of VND 50,000 per tree are imposed on illegal logging for a first-time offence but increase to an additional 2 million for each subsequent violation, with fines of VND 50,000–100,000 for each log. Fines for cattle grazing are VND 300,000 per incident, while government officers will destroy any implements or equipment used to damage mangroves. Furthermore, according to government agency staff interviewed in all provinces, current projects (both government and foreign funded) are impeded by a lack of clear and well-enforced monitoring and evaluation mechanisms, which leads to low local compliance. Around 60% of households surveyed claimed their project-related jobs were never terminated, even if they failed to deliver what was stipulated in their contracts, and local authorities do not enforce the law. Similarly, 50% of younger men in Thuong village claimed that although villagers are paid to conduct patrols two to three times a week, they invariably fail to do so. Also, penalties are low, and although the law requires violators to plant new mangrove forests, there is no enforcement of this. Moreover, although commune people's committees do issue local regulations prohibiting logging, according to one young member of a women's group interviewed in Bon village, community members still violate these regulations. Although FGD participants in Bon village said they were unaware of the government regulations, government officials said they had disseminated information about government policies, but local people were not complying with the rules.

DISCUSSION

A global review on the status of mangroves has shown that global mangrove dynamics are driven primarily by human and economic activities including pollution, over-extraction, and conversion of mangroves to aquaculture (Friess et al., 2019). This study's findings mirror this at both the national and provincial levels. Our study also echoes the global findings of Friess et al. (2019) where mangrove loss has slowed in recent years, and although some drivers, like firewood collection, are no longer a threat in our study sites, key drivers like aquaculture and infrastructure, which are rooted in national development strategies, continue to pose a threat to mangroves.

Our paper also shows that in similar fashion to other countries (Primavera and Esteban, 2008; Richards and Friess, 2016), despite large numbers of government and international conservation programs, for several reasons these have failed to address the drivers of mangrove deforestation successfully. This paper also shows the failure of many mangrove planting projects, and as **Figure 3** demonstrates, the area of mangroves in Vietnam has remained stable rather than increasing. Our findings demonstrate that both government and international projects only aim to address a sub-set of drivers (and often the easiest ones) while overlooking aquaculture as a key driver of deforestation (Dat and Yoshino, 2013; Pham et al., 2019b). Aquaculture is the key driver of mangrove loss not only in

Vietnam but in many other countries in the Asia Pacific region (Richards and Friess, 2016) and globally (Ahmed et al., 2018; Bosma et al., 2020). Without addressing this key driver of mangrove loss, pressure on mangroves cannot be mitigated. Our study also supports previous research showing that mangrove conservation interventions are not always effective because drivers of mangrove loss are often associated with national economic goals, like converting mangroves for aquaculture and agriculture production, and are not well addressed by policy interventions (Richards and Friess, 2016; Chowdhury et al., 2017; Thomas et al., 2017). The transformational changes required to address drivers effectively need to combine technical solutions and address the power relations and interests that drive forest loss (Gregorio et al., 2015; Moeliono et al., 2017). More specifically, they need to address the challenge posed by national and provincial aquaculture and development goals taking precedence over mangrove conservation, and national and international projects tending to overlook this.

Our findings show that surveyed households and FGD participants across the study villages commonly see government policies on expanding aquaculture and infrastructure as key drivers of deforestation and degradation. This suggests that addressing drivers of mangrove deforestation and degradation requires holistic national strategies. Our findings also show households, FGD participants and key informants perceiving the importance of each driver differently. While poverty and migration were cited in most FGD meetings as being major drivers of mangrove loss, only small numbers of key informants and surveyed households referred to these issues. This is probably because despite drivers of mangrove loss being common across sites, each village has its own socio-political context and hence some drivers are more site specific. This suggests that national mangrove conservation policies need to be adapted and tailored to take specific local conditions into account, and should consider different stakeholders' perceptions of existing problems and how they can be addressed.

This paper also reveals that policies and projects put strong emphasis on, and incentivize, the replanting of mangrove forests, but have not provided sufficient incentives for local people to protect existing mangrove forests. Earlier research has shown that existing forests can help countries to mitigate and adapt to climate change, but this is often overlooked in current forest protection mechanisms and financing, and therefore might threaten their long-term conservation potential (Funk et al., 2019). As our paper has shown, planting new forests might not always be successful. Like other countries, despite Vietnam receiving both government and international support to plant and protect mangroves, not all projects are successful due to low survival rates (e.g., only 10–20% in the Philippines) resulting from inappropriate site and species selection (Primavera and Esteban, 2008). Therefore, keeping existing forests healthy is essential (Overpeck and Breshears, 2021; University of Michigan, 2021). Future policies should promote both stabilizing existing mangrove forests and preventing mangrove deforestation.

Our paper also shows that most programs aim to pay for local labor costs as a part of reforesting or patrolling activities. However, these programs often lack sustainable financing (Jhaveri et al., 2018). Moreover, stakeholders have different views on how mangroves can be better protected in Vietnam (Hoang and Takeda, 2015). While government agencies see mangrove conservation as a technical matter, local communities see mangroves as providers of aquaculture—their main income source. Understanding local interests is key to designing well-implemented policies. To encourage people to plant mangroves, several authors have suggested combining monetary payments with in-kind support such as training and providing access to green markets, which as our study shows, were perceived by local people as more important than financial support. This is also because the payments offered by existing programs and projects are low and are unattractive to local people. FGD participants and household interviewees in all six villages said payments should at least equal potential earnings from other land uses. Many studies also show that support for local communities, such as establishing local small-scale fishery processing industries combined with ecotourism, clarifying land tenure and supporting forest land allocation to local communities, could help reduce poverty and enhance the effectiveness of mangrove conservation (Primavera, 2000; Santos et al., 2017). However, the sustainability of any scheme would also depend on sustainable finance.

Our paper also supports other studies in showing that the effectiveness of mangrove conservation programs depends on equitable benefit-sharing mechanisms and secured tenure where local people can feel their ownership over and responsibilities toward mangroves (Datta et al., 2012; Nguyen, 2014). Without secure rights and ownership, local people's commitment toward mangrove conservation will remain low. Our paper supports previous studies that have highlighted the critical importance of involving local people in participatory decision making (Walters, 2004; Datta et al., 2012; Santos et al., 2017) based on adequate understanding of and addressing existing power dynamics, full recognition of forest people's rights, and ensuring procedural, distributive and contextual equity (Pham et al., 2014b, 2021). However, the presence different actors (men, women and indigenous people) neither fully explains the process and dynamics of participation (Marochi, 2010) nor ensures successful policy and project outcomes (Cornwall, 2008; Vatn and Vedeld, 2013). The extent to which policies and projects can increase local participation in mangrove conservation also depends heavily on the capacities of local actors, as without the skills and resources to engage with local government structures, community participation can quickly become tokenistic (Hegga et al., 2020). Leadership capacity and elite capture, particularly at the local level, can undermine local participation in mangrove conservation (Agarwal, 2000, 2001, 2009; Pham et al., 2012; Kahsay and Medhin, 2020). Mangrove conservation policies also need to avoid "top-down" approaches with externally driven rules that fail to incorporate existing local norms and preferences (Orchard et al., 2015; Lau and Scales, 2016). We also found that local people view equitable benefit sharing and access to mangrove resources and information as important enabling conditions for their involvement and commitment to mangrove conservation. However, as we discovered, when only powerful and elite groups can access information and benefits from international and national programs, and forest resources are fully managed by government agencies (Hue and Scott, 2008; Hoang and Takeda, 2015), in the absence of other incentive mechanisms, this weakens interest in local communities to protect mangroves (Ha et al., 2012). Further, our paper shows that mangroves play an important role as a source of income for women, and women's unions can play an active role in mangrove conservation projects. Currently however, such projects limit opportunities for women to benefit, as schemes typically provide forest protection contracts that mostly pay men to patrol forests. Our paper also shares similar findings with a previous study by Hue, 2006 showing that the privatization of land in coastal areas would exclude the poor and women from accessing mangrove resources, which would greatly affect their incomes. Both men and women benefit from the conservation of mangroves (Bagsit and Jimenez, 2013; Treviño and Murillo-Sandoval, 2021), but amongst other factors, appropriate and equitable benefit-sharing mechanisms, and empowering women in policies and project design and implementation are required to enable women to take part in and benefit from mangrove conservation schemes (Barrero-Amórtegui and Maldonado, 2021). A gendered mangrove conservation approach is also essential for ensuring mangrove policies and projects understand and take into account both men's and women's interests and concerns (Feka et al., 2011; Bosold, 2012; Pearson et al., 2019) as well as ensuring good mangrove forest governance (Feka, 2015).

CONCLUSION

Our paper shows that mangroves in northern Vietnam have been destroyed as a result of economic development, local incomes from exploiting mangrove resources, clearing for aquaculture production, migration, poverty and water pollution. With an increasing understanding of the need to protect mangroves, the Government of Vietnam and foreign projects have put national and international policies and projects on mangrove conservation in place. There are no differences in the opinions of men and women or between different ethnicities in regard to drivers of mangrove deforestation and degradation, however, women are more vulnerable to the privatization of aquaculture production areas.

The effectiveness of mangrove conservation policies and projects depends on how well they can address drivers of mangrove deforestation and degradation. As these drivers are often driven by national development goals and other sectoral development needs with ministries competing for budgets and influence, holistic land-use planning needs to be coupled with effective coordination and clarification of responsibilities between government agencies, and coordinated and consistent policies concerning these natural resources. Addressing these underlying governance issues is far more important for mangrove conservation and restoration than merely offering financial incentives as various national and international projects have attempted.

In addition, policies need to pay attention to and incentivize both mangrove replanting and the conservation of exiting mangroves by providing technical support for planting and training on sustainable livelihood options for local people. Sustainable financing is essential, as are well-enforced policies, and accountable and transparent distribution of benefits and rights to stakeholders involved in mangrove protection. Gender sensitive policies and projects that take gendered differences into account in their policy interventions are important to ensure good governance. Previous projects can offer lessons for future programs, particularly on how to align local people's interests with the intended objectives of mangrove conservation projects, and on the effective use of both incentive and disincentive mechanisms.

DATA AVAILABILITY STATEMENT

The data are not publicly available as they contain information that could compromise research participant privacy/consent.

ETHICS STATEMENT

Ethical review was approved by local government authorities which are Department of Agriculture and Rural Development of Thanh Hoa Province, Forest Protection Department of Thanh Hoa Province, Forestry Department of Thanh Hoa Province, Forest Protection and Development Fund of Thanh Hoa Province, Department of Agriculture and Rural Development of Thai Binh Province, Forest Protection Department of Thai Binh Province, Forest Protection Department of Quang Ninh Province, and Tien Yen Protection Forest Management Board and Dong Rui Commune People Committee. The participants provided verbal informed consent to participate in this study.

AUTHOR CONTRIBUTIONS

TP: conceptualization, methodology, formal analysis, resources, writing—original draft, and supervision. TV: conceptualization, methodology, validation, resources, editing, and supervision. TH and TD: investigation, formal analysis, data curation, and visualization. DN, DP, LD, and NH: investigation. VN: investigation, software, and data curation. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/ffgc.2021.709073/full#supplementary-material

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Mangroves From Rainy to Desert Climates: Baseline Data to Assess Future Changes and Drivers in Colombia

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INTRODUCTION

Mangroves in Colombia (northern South America) have been described as the most luxuriant and wettest of the Americas (West, 1956) and as carbon-rich-tall tidal forests (Hutchison et al., 2014; Hamilton and Friess, 2018; Simard et al., 2019; Castellanos-Galindo et al., 2021a,b). Such statements mostly come from expeditions and research conducted on the Pacific coast and from global studies emphasizing on its ecological significance (e.g., Polidoro et al., 2010). However, mangroves in Colombia also occur along the Caribbean coast and in an oceanic archipelago off Nicaragua in Central America (Blanco-Libreros and Álvarez-León, 2019). Such distribution comprising mainland and oceanic settings allows for mangroves in the Colombian territory to exhibit unique features as a result of the wide variety of biogeographical regions and climates. First, mangroves are located in three biogeographical regions: the Tropical Eastern Pacific, the Southern Caribbean, and the oceanic Caribbean (i.e., San Andrés, Old Providence, and Santa Catalina Islands Archipelago; García-Hansen et al., 2002; Medina-Calderón et al., 2021). Out of the 285,040 ha, 194,880 ha are located in the Pacific region and 90,160 ha in the Caribbean region (including the oceanic territory). It is noteworthy, that the disconnection of the ancient coast of northern South America after the rise of the Panama Isthmus induced the disjunct distribution of *Pelliciera*, the only endemic genus to the New World (Duke, 2020), absent from oceanic Caribbean (García-Hansen et al., 2002; Medina-Calderón et al., 2021). Second, mangroves exist from super-humid (>7,000 mm y^{-1}; Castellanos-Galindo et al., 2017; Riascos et al., 2018) to desert climates (<500 mm y^{-1}; Guajira Peninsula), a fact little acknowledged in the literature addressing either rainfall gradients or arid zones (e.g., Osland et al., 2018; Adame et al., 2021). Moreover, mangroves are found throughout a wide range of geomorphic settings such as large to small deltas, estuarine, lagoons, open coasts, and carbonate islands (following the classification by Worthington et al., 2020). Finally, there are marked contrasts between the Pacific and Caribbean biogeographical regions relative to land use and land cover, particularly urbanization (Blanco-Libreros and Ramírez-Ruíz, 2021).

The unique settings of mangroves in Colombia provide a remarkable opportunity to study biogeographical, macroecological, regional, and landscape-level patterns of species composition, forest structure, and ecosystem function, as well as the anthropogenic and natural drivers of spatiotemporal change. In particular, hydrological alterations due to human activities and land use change seem to be the main drivers along the Caribbean coast of Colombia (Ward et al., 2016; Jaramillo et al., 2018; Blanco-Libreros and Ramírez-Ruíz, 2021). In addition, focusing on climate change impacts on mangroves, recent global studies suggest contrasting patterns between the Caribbean and Pacific coasts of South America, particularly related to sea-level rise, storminess, altered precipitation regimes, and erosion (Ward et al., 2016; Goldberg et al., 2020). Consequently, open-access, large-scale databases and baseline information are urgently needed to understand the spatiotemporal patterns of change and drivers in Colombia. However, a major challenge is to deal with the difference in sampling periods and methods by different surveys, and it is thus very important to assemble data obtained with standardized methods or obtained during a single study (e.g., Kauffman et al., 2020).

Accordingly, in previous work, we curated a database obtained during the major mangrove survey available to date in Colombia, conducted by the Ministry of the Environment (HELIO_SP.CO v.1; Blanco-Libreros and Álvarez-León, 2019). The survey was conducted in the mid-1990s, but no similar effort has been undertaken afterward. Subnational or departmental (administrative level 2) surveys have been carried out since year 2000 but reports and data are not easily accessible (**Supplementary Material**). Moreover, mangrove research comprising regional and national levels are scant in Colombia due to limited funding and complex logistics (reviewed by Castellanos-Galindo et al., 2021a), leading to a situation where only a few highly committed researchers have been able to gradually expand the geographic coverage of their research programs (e.g., Polanía et al., 2015; Urrego et al., 2018).

Here, we update the aforementioned database in response to requests made by colleagues for assembling mangrove forest structure datasets at national and sub-national levels. Such datasets are useful for estimating and modeling blue carbon, fine-tuning global models, and validating national and global mangrove maps (Rovai et al., 2016, 2021a,b; Bolívar et al., 2018; Hamilton and Friess, 2018; Mejía-Rentería et al., 2018; Simard et al., 2019). Analysis of datasets over broad extents has allowed recent progress of mangrove macroecology, particularly concerning blue carbon (e.g., Rovai et al., 2016, 2021a; Macreadie et al., 2019; Sasmito et al., 2019).

In HELIO_SP.CO v.1, we only included mainland locations (1.41–12.23 N; 71.28–79.00 W), while in this updated version we added locations in the oceanic Caribbean (12.32 N, 81.41 W, García-Hansen et al., 2002). The oceanic Caribbean was surveyed alongside with mainland locations, using the same methods, but only summary data were included in the original report (see Methods). This is a timely update because San Andrés and Old Providence islands (the largest within this territory and the only sustaining mangrove patches) were hit by hurricanes Eta and Iota in November 2020 (Garcés-Ordoñez et al., 2021). On that ground, the inclusion of historical data will also serve as a baseline for ecological impact assessments and ecosystem modeling of succession trajectories. In addition, we expanded the number of forest-structure attributes for the entire database to seven variables for five species (see Methods). All variables appeared in the original reports, but we only included tree density, mean tree diameter at breast height (dbh) and Importance Value Index (IVI). Mean tree height was only reported for locations on the Caribbean coast. The main objective of the HELIO_SP.CO v.2 is to serve as a baseline to assess changes in mangrove species occurrence, forest structure attributes, and coastalscape features after 2000. This database can serve as an input for future analyses to better understand the anthropogenic and natural drivers, to support decision-making in conservation and restoration, and to support climate change mitigation strategies.

METHODS AND DESCRIPTION OF THE DATABASE

HELIO_SP.CO v.2 (**Supplementary File**) comprises mangrove inventory data for 113 locations (11 more than in version 1) covering a broad variety of physiognomic types under contrasting climates and geomorphic settings (**Figure 1** and **Supplementary Figure 1**). Each location corresponds to a georeferenced entry-point (Garmin 5 GPS, WGS 84 System, with a 100-m precision) in the mangrove fringe where the forest inventory was carried out. This database reports tree density, mean tree diameter at breast height (dbh, measured at 1.3 m above ground, with modifications depending of height of aerial roots and branching patterns), basal area, frequency, relative density, dominance and importance value index for *Rhizophora* spp., *Avicennia germinans*, *Laguncularia racemosa*, *Pelliciera rhizophorae* and *Conocarpus erectus*. For the Caribbean it has well stablished the presence of *R. mangle*, but for the Pacific, this species coexists with *R. harrisonii* and *R. racemosa*, therefore the report for this coast referred to the species complex as *Rhizophora spp*. The original forest inventory was carried out following either the Point-Centered Quadrat Method (20 m diameter) or the Alternated Square Plots (25 m^2), but all data were originally reported upon 0.1 ha. Mangroves were sampled at least in 10 points along a transect perpendicularly to the shoreline. Trees sampled within each quadrat or plot were divided into three diameter categories (>15, 5.1–15, and 1–5 cm). We only included dbh > 15 cm because they can be more representative of long-term trends or steady-state canopy conditions (see discussions on large tree inventories in terrestrial forests:). In addition, small-diameter trees (1–5 cm) were not measured in the Pacific coast. Fieldwork was carried out simultaneously in both coasts and the oceanic Caribbean between November 1995 and August 1996. Further descriptions are available in the printed reports by Saìnchez-Paìez et al. (1997a,b) cited by Blanco-Libreros and Álvarez-León (2019); **Supplementary References**).

Data from San Andrés Island were not included in version 1 of the database because the printed volume for the Caribbean did not include summary tables and it just described major features while the data mentioned in the text referred to an unpublished honors thesis. Since geographic coordinates were not reported in text for the sampling sites in San Andrés, we estimated the proximate coordinates for 11 sites by comparing the printed map included in the report and in additional publications (García-Hansen et al., 2002; **Supplementary References**) with Google Earth Pro. We further cross-checked the estimated location

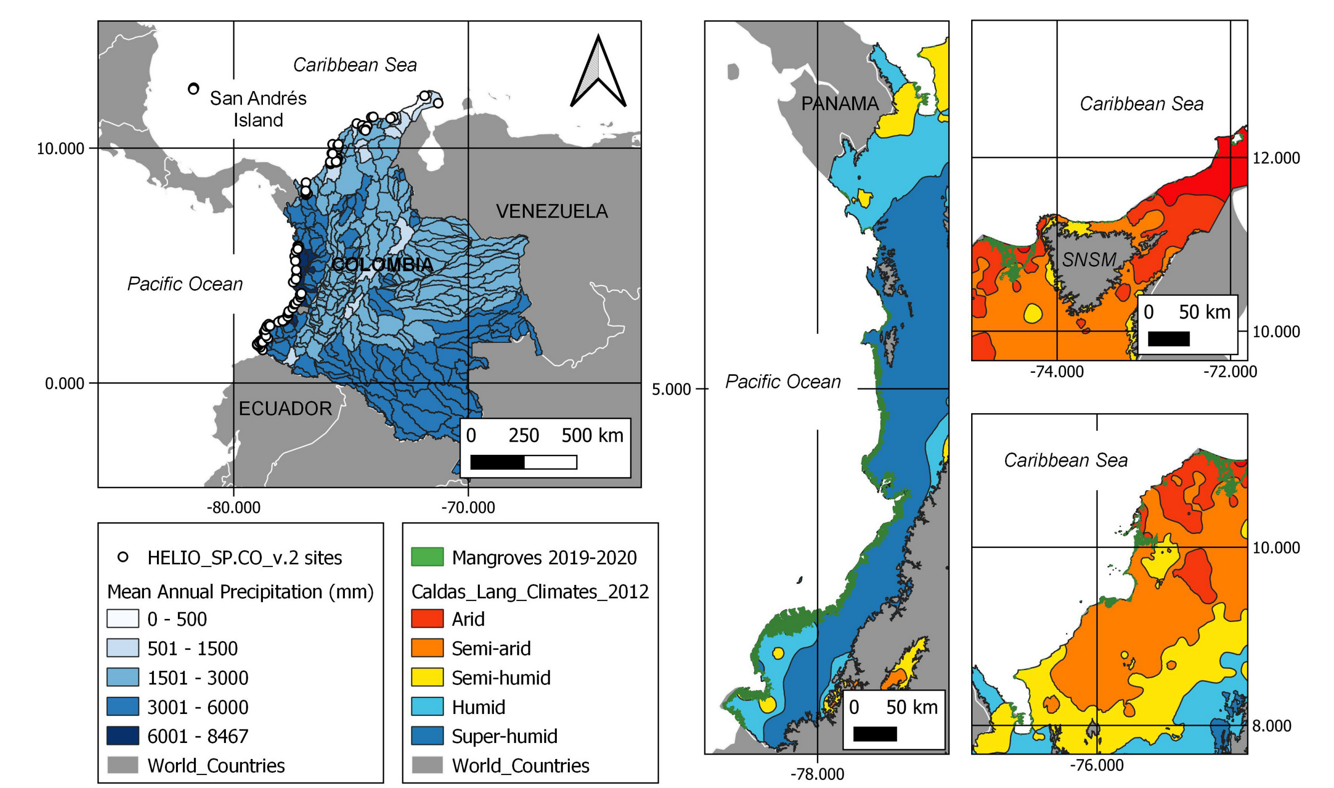

FIGURE 1 | Sites included in HELIO_SP.CO v.2 database are shown relative to mean annual precipitation within watersheds (left: note the location of San Andrés Island, oceanic territory in the Caribbean Sea-OC). Climates according to Caldas-Lang classification are shown for the entire Pacific coast and northern and southern stretches of the continental Caribbean coast (center and right). Sierra Nevada de Santa Marta (SNSM: a major coastal massif in the northern Caribbean coastal plain), Western Cordillera (one of the branches of the Northern Andes), and Serranía del Baudó (a satellite system running parallel to the northern Pacific coast) were excluded from the representation of climates in the insets (shown in gray color). Climate data from IDEAM, and mangrove map (2019–2020) from Valencia-Palacios and Blanco-Libreros (2021). Drawn with QGIS Zürich.

with the current official mangrove cover map (**Supplementary Figure 2**).[1] The senior author visited the sites in September 2021 to inspect anthropogenic changes, and concluded that conservation efforts have maintained mangrove extent and ecological conditions similar to the descriptions by García-Hansen et al. (2002). Maps and descriptive statistics were included as **Supplementary Material**.

BRIEF ANALYSIS

Country-Wide Patterns

The current database includes 61 sites along the Pacific coast under superhumid (total annual precipitation, $P > 5{,}000$ mm y^{-1}) and humid ($P > 2{,}500$ mm y^{-1}) climates, and 52 sites along the Caribbean coasts (including oceanic and continental areas) extending over semi-humid (in Antioquia, Southwestern Caribbean), semi-arid and arid (Cordoba-Magdalena), and desert (Guajira, Northeastern Caribbean) climates (P range: < 500–$< 2{,}500$ mm y^{-1}) (**Supplementary Figure 3** and **Supplementary Table 1**). Country-wide, *Rhizophora* spp. exhibited the greatest IVI (mean: 165.7; range: 0–300) given the high relative density, dominance and frequency, in continental (CC) and oceanic (OC) locations in the Caribbean (means: 89.7 and 134.4, respectively), but most significantly along the Pacific (222.7) (**Supplementary Figures 3,4**). *Avicennia germinans* was the second-most important species (mean: 27.3; range: 0–195), with the higher contribution in the OC and CC (even forming monospecific fringes in basin settings, **Supplementary Figure 1**) than in the Pacific (means: 81.3, 36.1 and 11.7, respectively). The third species was *Laguncularia racemosa* (mean: 19.2; range: 0–205) with greatest values in the OC, followed by CC and the Pacific (means: 85.6, 22.9 and 4.7, respectively), seemly as a response to the natural and anthropogenic disturbances. *Pelliciera rhizophorae* is scant and of little importance country-wide, but forms monospecific stands in some areas along the Pacific coast. *Conocarpus erectus* was recorded in a few sites along the Caribbean and the Pacific coasts, mostly due to its habit of colonizing the inner-most areas with well-drained sediments.

We previously reported that IVI for *Rhizophora* and *Avicennia* was partially correlated with mean annual temperature, mean annual rainfall, and rainfall seasonality (collating data from WorldClim 2; Blanco-Libreros and Álvarez-León, 2019), therefore clear differences in species composition and metrics

[1]http://sigma.invemar.org.co

are expected among Colombia's climatic zones. Using the current database, the correlation of IVI with latitude and longitude coordinates (**Supplementary Figure 5**) shows marked biogeographical patterns. For instance, *Rhizophora* displayed a negative correlation with both latitude and longitude, while *A. germinans* showed a positive correlation with latitude. *L. racemosa* and *P. rhizophorae* exhibited positive and negative correlations with latitude, respectively. Stronger patterns would be expected when crossing HELIO_SP.CO v.2 data with Caldas-Lang climate and total annual precipitation data (**Figure 1**), as well as other sources (e.g., Koppen-Geiger climate classification). The addition of OC data to the database provides an opportunity to explore the influence of dry and maritime climate (influenced by cyclonic activity) on mangrove attributes.

We recommend using structural variables such as density, mean tree diameter, and basal area for understanding the climatic and biotic drivers of macroecological patterns. For instance, by using crossed linear correlations between density for the three dominant species (*Rhizophora* spp. *A. germinans* and *L. racemosa*) and geographic coordinates, insights are gained about the overall role of climate on large-scale distributions and ecological interactions at plot or local scales (**Figure 2**). Density in *Rhizophora* spp. was negatively correlated with latitude and longitude country-wide, indicating the negative effect of reduced rainfall, particularly along the CC. Density in *A. germinans* showed no significant correlation but it increased significantly to the northeastern coast of the CC (toward arid and desert climates). As a consequence, both species showed an antagonistic pattern country-wide but more markedly along the Caribbean region. Along this coast in particular, *R. mangle* forms extensive monospecific stands toward the southwest and *A. germinans* does it toward the northeast. In the Central Caribbean where both species coexist, they are spatially segregated, with *R. mangle* forming monospecific seaward fringes and *A. germinans* forming monospecific basin stands. Such patterns demonstrate the differences in ecological niches described in the literature. Finally, *L. racemosa*, a species that has not been studied in depth in the literature, shows significant correlations with latitude and longitude at different spatial scales and with *A. germinans*. Such patterns suggest a complex interaction between climatic, biotic and disturbance regime drivers. For instance, while country-wide *L. racemosa* is positively correlated with *A. germinans*, it is negatively correlated along the CC. This might result from the out-competition by *Rhizophora* spp. along the Pacific coast, but the release of local competition with *A. germinans* in the Caribbean coast with increased disturbance due to logging or cyclonic activity (e.g., Blanco-Libreros and Estrada-Urrea, 2015). Spatial patterns for *P. rhizophorae* and *C. erectus* are weaker due to the scarcity of occurrences country-wide (**Supplementary Figure 6**). However, the use of HELIO_SP.CO v.2 data in combination with other sources of information may be helpful for studying such patterns [see Blanco-Libreros and Ramírez-Ruíz (2021)]. We recommend the exploration of interactions among climatic, geomorphic and biotic drivers by using multiple regression models (both linear and non-linear, geographically structured or not). Finally, we also recommend studying how the disturbance regime in OC seems to promote species coexistence among the three dominant species island-wide in San Andrés and how they are locally segregated [see Medina-Calderón et al. (2021)].

Potential Uses

We propose four main potential uses of the present database: (1) species distribution modeling and species conservation status assessment, (2) above-ground blue carbon and ecosystem services estimation, (3) land cover and use assessments in the coastalscape, and (4) current and future responses to coastal climate change. Presence and absence data can be used in large-scale species distribution modeling (SDM) efforts using various mathematical approaches, and they have been recently applied to neotropical mangroves (Rodríguez-Medina et al., 2020). WorldClim, CHELSA, and other climate data sources are commonly used for SDM. Presence and absence data, in combination with records in the Global Biodiversity Information Facility (GBIF), can also be used as spatial references for designating areas of interest (buffers) to study threats to vulnerable species such as *Pelliciera* spp. (see Blanco-Libreros and Ramírez-Ruíz, 2021). For this species, we used landscape metrics to understand mangrove habitat fragmentation relative to urbanization, quantifying the magnitude of this specific threat, an approach that can be applied to other mangrove tree species.

Second, plot-level mean density and mean tree diameter are useful for estimating carbon in the above-ground biomass (AGC) using allometric equations (e.g., Yepes et al., 2016; Rovai et al., 2021b). Given the scarcity of field studies in Colombia, the present database may help to reduce spatial uncertainty in national and sub-national level modeling efforts (Bolívar et al., 2018). We have estimated that plot-level AGC may range between < 10 and 225 Mg ha^{-1} in the mainland (G. F. Pérez-Vega unpublished monograph, **Supplementary Figure 7** and **Supplementary References**). In addition to the climate mitigation ecosystem service provided by AGC, the basal area is correlated with wave dissipation capacity, functioning as coastal protection (Sánchez-Nuñez et al., 2020). Basal area (particularly in *Rhizophora mangle*) in this database may help to assess such service in OC and CC areas seasonally impacted by storm surges.

Third, land cover and land use change within mangroves and in its surroundings is a major driver of deforestation and fragmentation worldwide (Bryan-Brown et al., 2020). Species occurrences and forest attributes can be used as response variables to past drivers or as baseline information to assess future change. A recent assessment of the coastalscape configuration around *Pelliciera* spp. occurrences in response to urbanization may serve as an example (Blanco-Libreros and Ramírez-Ruíz, 2021). We encourage the use of the recently published official land cover data[2] to assess the current state of the coastalscape and to use urban expansion data[3] to understand threats to urban mangroves in the largest coastal cities in Colombia (e.g., Cartagena, Buenaventura). Finally, global layers such as the

[2]http://www.ideam.gov.co/web/ecosistemas/coberturas-nacionales
[3]https://marroninstitute.nyu.edu/blog/urban-expansion-work-in-colombia

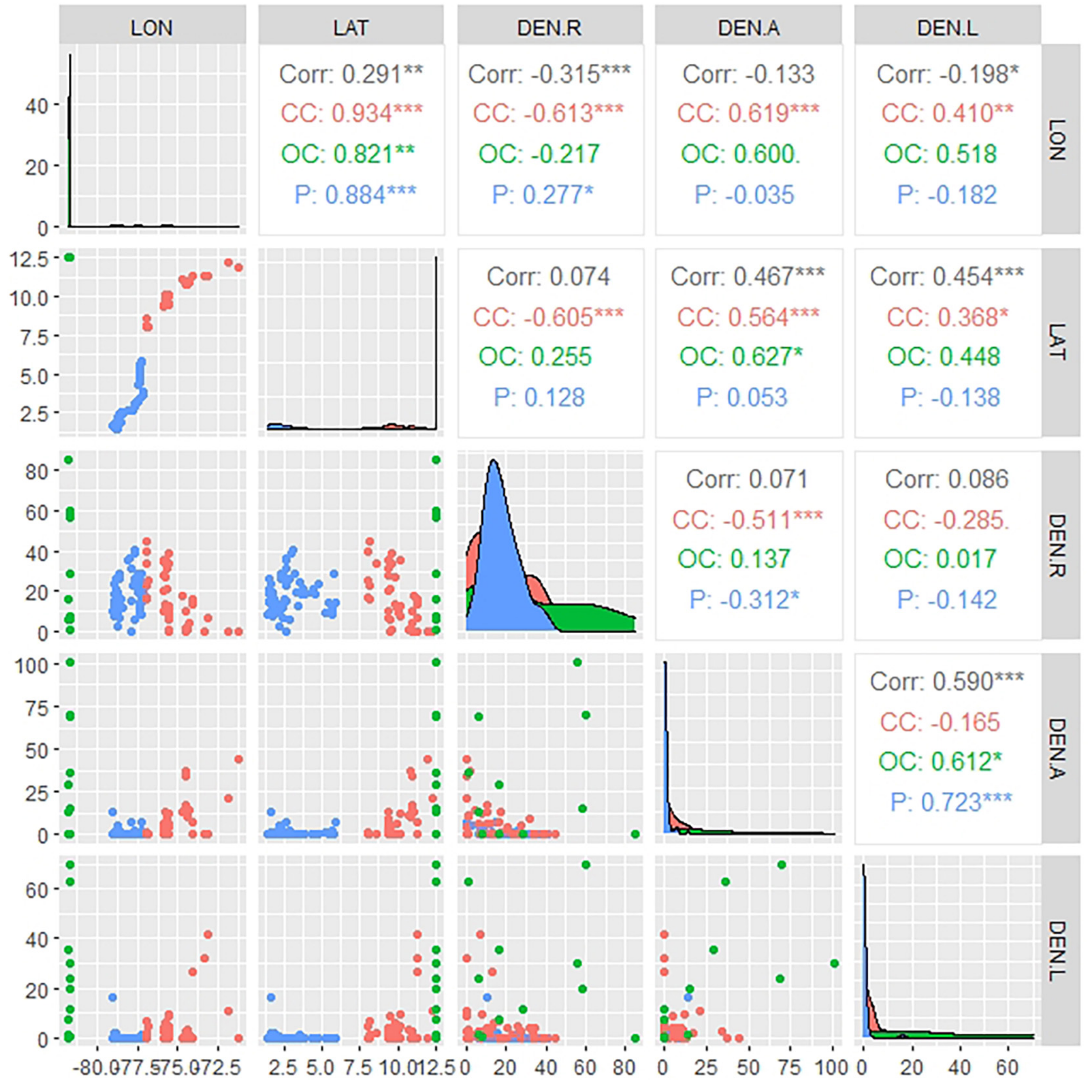

FIGURE 2 | Correlation matrix between the geographical coordinates and density values for the three dominant mangrove species (*Rhizophora* species complex, *Avicennia germinans*, *Laguncularia racemosa*) in the continental and oceanic coasts in the Caribbean (CC and OC, respectively) and the continental Pacific coast (P) of Colombia (note color coding). The upper half of the matrix shows the Pearson correlation indices with significance level indicated as asterisks for the total dataset and subsets for CC, OC, and P. The lower half panels show the scatter plots, and the diagonal boxes show the frequency distributions for each coast in different colors. Data available in HELIO_SP.CO v.2. Drawn using *ggpairs* function in *GGally* extension for *ggplot2* package for R.

nighttime lights,[4] and national layers of national parks, african-descendant territories and coastal watersheds,[5] and demographic variables for departments and municipalities[6] can be also useful to understand the spatial coastalscape context and socio-economic dynamics affecting variables in mangroves in Colombia (**Supplementary Figure 8**). A few studies of this type have been conducted in mangroves and terrestrial forests in the Chocó-Darién Ecoregion (e.g., López-Angarita et al., 2018; Fagua et al., 2019). Land cover and land use change have been anecdotally suggested as major drivers of changes in mangrove area and species composition along the Caribbean coast of Colombia but a few quantitative studies exist (e.g., Blanco-Libreros and Estrada-Urrea, 2015; Mira et al., 2019; Bolívar-Anillo et al., 2020; Villate-Daza et al., 2020). Therefore, we strongly encourage colleagues to use the present database to advance in this field.

Climate change and climate variability have been described mostly along the Caribbean coast of Colombia.[7] Future reductions in annual rainfall are predicted for the mid and northern Caribbean. Increased water stress on coastal watersheds is predicted for some areas due to land use changes, particularly urbanization. In addition, although the El Niño-Southern Oscillation is a strong driver of interannual variability on rainfall

[4]https://blackmarble.gsfc.nasa.gov/
[5]http://www.siac.gov.co/
[6]https://geoportal.dane.gov.co/geovisores/
[7]http://www.siac.gov.co/cclimatico

and runoff in Colombia its influence on Colombian mangroves has been little studied (i.e., Galeano et al., 2017; Riascos et al., 2018; Riascos and Blanco-libreros, 2019). However, strong El Niño and La Niña events are seemly responsible for species composition transitions in the mid-Caribbean coast (Bolívar-Anillo et al., 2020; Villate-Daza et al., 2020). In the specific case of San Andrés and Providence islands, as oceanic territories, an increased rate of cyclone activity is expected. Hurricanes Eta and Iota hit both islands on November 2020 affecting mangroves (Garcés-Ordoñez et al., 2021). The prevalence of such climatic and meteorologic drivers urges for the need for baseline data and for setting local, regional, and national monitoring programs. Nowadays, a long-term monitoring program only exists for Ciénaga Grande de Santa Marta (see text footnote 1). Additional permanent plots have been established in some departments of Colombia but maintenance and repeated measurements are contingent on budget constraints producing many gaps in the records or even abandonment of the monitoring programs. Long-term studies and datasets are also needed for understanding the role of oceanic drivers such as coastal erosion and sea level rise.

Finally, we are certain that HELIO_SP.CO v.2 data are also useful for validation of mangrove maps. The Ministry of Environment and Sustainable Development issued in 2018 the Decree 1,263 ushering the environmental departmental authorities to update mangrove maps by 2021. The validation of such maps requires field campaigns over extensive areas of difficult access. In order to contribute to the advance in this task, we successfully used version 1 coordinates as validation points for a 2019–2020 map built using Sentinel 2 imagery and cloud computing in Google Earth Engine [**Supplementary Figure 8**; see data in Valencia-Palacios and Blanco-Libreros (2021)]. Thus, we encourage mangrove cartographers to use the present database as an alternative validation way given the growing use of cloud-based mapping in Colombia, particularly in areas of remote access making large-scale ground-truthing either logistically difficult or prohibitively costly (e.g., Perea-Ardila et al., 2021). We also foresee applications for estimating anthropogenic pressures relative to distance from large populated centers, as well for estimating the benefits perceived from mangrove-based fisheries (**Supplementary Figures 9,10**). Despite departmental-level forest inventories have been conducted since 2000, official data are not easily disclosed to scientists (**Supplementary Figure 11**), and scientific forest structure studies published since 2000 still have a very limited geographical coverage [discussed by Bolívar et al. (2018) and Castellanos-Galindo et al. (2021a)]. We conclude that HELIO_SP.CO v.2 can be useful as a baseline for the XXI century to assess future change and drivers in Colombian mangroves.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author/s. HELIO_SP.CO v2 data are freely available at Harvard Dataverse: https://dataverse.harvard.edu/dataset.xhtml?persistentId=doi:10.7910/DVN/QXQT59.

AUTHOR CONTRIBUTIONS

JB-L and RÁ-L conceived, more than 20 years ago, the use of this database by a broader audience and finally published the first version and conceived the present update. JB-L supervised data entry and lead curation. JB-L and SL-R wrote the first draft with input from RÁ-L. AV-P and GP-V helped with database curation, ran exploratory statistical procedures, built exploratory maps, analyzed the data, contributed ideas and procedures for potential uses, and contributed to manuscript writing. All authors provided input and approved the submission of the final manuscript.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/ffgc.2022.772271/full#supplementary-material

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Tangled Roots and Murky Waters: Piecing Together Panama's Mangrove Policy Puzzle

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Mangrove forest policies are often characterized by their fragmented nature, as multiple sectors, disciplines, and institutional structures interact to affect mangrove conservation and management. This study analyzes mangrove forest policies in Panama, a country known for its rich mangrove coverage and, conversely, its high rates of mangrove loss, urban expansion, and coastal development. To complement the policy analysis, key informant interviews with national policy actors are used to gather insights on policy implementation challenges and potential multi-actor collaboration opportunities. Results suggest that despite the development of multiple policies targeting wetlands and conferring a high conservation status to mangroves in Panama, mangrove protection is challenged by competing governmental agendas and policy implementation gaps. Efforts to strengthen mangrove conservation and initiate participatory management processes were also found to conflict with institutional structures that struggle to include local communities and foster collective action.

Keywords: mangrove forest policy, sustainable mangrove management, Latin America, collaborative governance, policy analysis

INTRODUCTION

As outlined by many global conservation reports, mangroves are one of Earth's most productive, resilient, and biodiverse ecosystems, but also one of the most poorly protected (Van Lavieren et al., 2012; Duke et al., 2014; Slobodian and Badoz, 2019). Mangroves are often a "blind spot" in environmental policy because they cross multiple boundaries, partly coastal habitats, forests, and tropical wetlands. Despite their unique ecology, few countries have passed a law specifically designed for mangroves (Spalding et al., 2010). Instead, many of the national legal regimes governing mangrove ecosystems are fragmented and complex (Slobodian and Badoz, 2019). For example, mangroves are often managed under legal frameworks intended for the environment, forests, water, wetlands, and fisheries, which can fall under many governmental jurisdictions and sectoral responsibilities (Rotich et al., 2016). Policy tools to protect mangroves can take multiple forms, such as direct protection of mangrove species, protected areas, logging permits, Environmental Impact Assessments, integrated land-use planning, and collaborative management approaches—including Indigenous-led management (Friess et al., 2016; Slobodian and Badoz, 2019). Although many policy tools exist, legal effectiveness and compliance with mangrove policies are often found to be deficient, leading to accelerated mangrove loss (Rotich et al., 2016).

As the Central American country with the largest area of mangrove cover, Panama is a compelling site to study mangrove management policies (Spalding et al., 2010). A large extent of Panama's mangroves are included in the National System of Protected Areas (approximately 70,000 hectares), but mangrove coverage continues to decline rapidly (Tarté, 2013). Panama has lost at least 13 percent of its mangrove cover between 1996 and 2008 due to its growing infrastructure sector, among other reasons (Dow, 2008; Tarté, 2013; López-Angarita et al., 2016). The greatest proportion of mangrove destruction in Panama has occurred around Panama City, where the space occupied by wetlands competes against numerous projects of urban expansion (Kaufmann and Miró, 2012). Mangroves' prized location on coastal lands with high economic value generates pressure on their conversion due to other land uses (e.g., aquaculture, commercial, industrial, residential, ports) (Spalding et al., 2010). Aside from habitat loss, mangroves in Panama also face degradation from nutrient runoff, water contamination, and extreme weather events (Lin and Dushoff, 2004; Defew et al., 2005; Tarté, 2013).

Previous reviews of Panama's coastal and mangrove management policies have found that coastal policies face several administrative and structural gaps (Osorio, 1994; Suman, 2002; Spalding et al., 2015). For example, since the 1990s, a major institutional reorganization has been occurring in Panama, leading to coastal management responsibilities becoming fragmented among many government agencies. Interagency coordination is deficient in Panama with no formal coordination mechanism and limited cohesive vision appearing in coastal contexts (Suman, 2002; Spalding et al., 2015). Existing policies have been found to aggravate insecure property rights on Panama's coasts, as local communities struggle to own land and to secure access to coastal resources and economic activities such as fisheries and tourism (Spalding et al., 2015). Building on this work, our study aims to understand the extent to which national policies and legal instruments foster sustainable mangrove management in Panama and how responsible authorities coordinate mangrove management. Sustainable mangrove management can be defined as the inclusive application of practices that "will help to achieve multiple objectives of management and utilization of [mangrove] products without any undesirable effect on the physical and social environment" (Datta et al., 2010, p. 468), requiring "effective and accountable governance and the safeguarding of the rights of forest-dependent peoples" (Blaser, 2016, p. 11).

MATERIALS AND METHODS

We employ a hybrid approach combining policy analysis and a thematic analysis of qualitative interviews, following the combined approach by Spalding et al. (2015). Our approach had a "strategic" orientation (Srivastava and Thomson, 2009), as we identified avenues for policy improvement and/or new strategies, especially at the stage of policy implementation. For the policy analysis, we reviewed all existing policies related to mangroves at the national level in Panama, excluding soft laws and municipal council decisions. Data for this review were obtained by scanning the scientific literature and soliciting legal documents from government offices. Further, many mangrove policies were obtained *via* searches in InfoJurídica, a Panamanian legal database comprising detailed expression of laws' impacts, validity, and unconstitutionality (Infojurídica, 2020).

To complement the information contained in policy documents, we conducted eight semi-structured interviews lasting approximately 45 minutes with key informants involved in mangrove policy and management in Panama, including scholars, policymakers, members of mangrove advisory bodies, and non-governmental organization (NGO) experts (see **Table 1**). These actors were purposively selected based on their involvement in national mangrove policymaking and their participation in mangrove management groups, such as the National Committee on Wetlands (Palinkas et al., 2015). Interviews allowed us to gather deeper insights to the stakeholder interactions supporting policy objectives, in addition to better understanding the application of existing mangrove-related policies. Interviews were conducted in Spanish and English in Panama between February and March 2020, and online through password-protected video-conferencing platforms in April 2020. Interview question guides are available as **Supplementary Material**. All field research protocols were reviewed and approved by the McGill University Research Ethics Board (REB File #: 19-11-046) prior to data collection. The project also received a local research permit from the Smithsonian Tropical Research Institute (STRI) (Protocol # HS20005).

Interviews were fully transcribed and analyzed in MaxQDA, a qualitative analysis software that enabled data to be classified (coded) into themes (Guest et al., 2011). Coding of recurring themes was performed manually following an inductive-deductive approach (Fereday and Muir-Cochrane, 2006). Deductive reasoning was used to build a coding matrix based on recurrent mangrove policy challenges identified by Friess et al. (2016) and by A. K. Spalding et al. (2015), yielding the following broad themes: conflicting policy objectives, overlapping jurisdictions, implementation of protected areas, collaborative governance, increased role of the private sector in management, and coastal property regimes. To identify any relevant additional themes, we complemented our analysis with inductive reasoning, where new themes and categories emerged directly from the data through careful examination and constant comparison of interview transcripts and policies (Memon et al., 2017).

Small sample size is a limitation of this study, however is considered adequate because of the study's design, the presence of key informants, and the scoping intent of interviews. According to the "information power model," our study's design is compatible with smaller samples sizes because the selection of participants is highly specific to the study's aim and the interview dialog is strong (Malterud et al., 2016). In addition, interviews were mainly used to identify and scope potential mangrove management issues to be assessed in-depth in subsequent studies. Content validity in this study was ensured by conducting a pre-test of the interview guide with non-participating stakeholders and by using peer-reviewed frameworks on mangrove management (Brod et al., 2009). In addition, we employed a triangulation strategy in our research

TABLE 1 | Overview of interviewed stakeholders.

Stakeholder group	Informant's organization	Organization's role	Distribution per stakeholder group
Government	Ministry of Environment (various divisions) ARAP	Develop policies on the management of mangrove forests, manage protected areas, and allocate funding to projects in mangrove forests (restoration, education, protection) Manage the impact of mangrove forests on artisanal fisheries, and payments for mangrove clearing	4
Scientists	Smithsonian Tropical Research Institute International Maritime University of Panama	Provide scientific insight on mangrove management and policy	2
Non-governmental organizations	Centro de Incidencia Ambiental de Panamá (CIAM) Sociedad Audubon de Panamá	Support environmental protection and conduct strategic litigation Support the protection of mangrove ecosystems through community engagement and government partnerships to support bird populations	2
TOTAL			8

protocol by combining different methods (semi-structured interviews, participant observation, and documentary analysis) and by encouraging the participation of stakeholders from diverse backgrounds (Baxter and Eyles, 1997).

RESULTS

The Legal Framework of Mangrove Protection in Panama and Its Recent Developments

Even though Panama does not possess a law specifically designed for mangroves, many laws are used to govern mangroves and tropical wetlands (see **Table 2** for a list and description of relevant laws and policies). At first glance, Panama's laws and policies appear to support the preservation and sustainable management of natural resources; the most striking example of this being the strong environmental protection language used in the 1972 Constitution. Within the General Law for the Environment (Asamblea Legislativa, 1998), which regulates the use of natural resources and promotes the pursuit of environmental preservation, mangroves are given a high conservation priority. In 1989, Panama engaged in further steps to protect mangroves and other wetlands by signing the Ramsar Convention on Wetlands of International Importance. The implementation of the Convention was supported by the creation of the National Committee on Wetlands in 2007 and the Política Nacional de Humedales in 2018 (National Wetlands Policy) (Ministerio de Ambiente and the United Nations Development Programme [PNUD], 2018). Yet these efforts have been undermined by weak compliance with mangrove protection standards, as highlighted by our interviews. For example:

> "*Panama is a country where there are enough, if not too many laws. There are laws for everything, for everyone. But the problem is, in my opinion, compliance with these laws. There are many, many laws, but they are not enforced. [...] Whether I am a businessman with a lot of money or a common citizen, I must develop a feeling that I must comply with the law because otherwise I will be punished by the authorities*" (Scientist).

Mangrove protection policies are overshadowed by the dominant mangrove policy created in 2008 by the Autoridad de los Recursos Acuáticos de Panamá (ARAP), Panama's Authority on Aquatic Resources. Through Environmental Impact Assessments (EIA), ARAP requires authorization for any activities affecting mangrove ecosystems (Autoridad de los Recursos Acuáticos de Panamá, 2008a). However, key informants reported that EIA is "a formality" that often leads to approval of development projects (tourism, industry, and ports) occurring in mangrove habitat, to the point where land conversion is cited as the main source of mangrove loss in Panama (Kaufmann and Miró, 2012; López-Angarita et al., 2016).

The evolution of ARAP policies related to mangrove logging and deforestation support this claim, revealing a discounting of mangrove benefits. Early ARAP resolutions (Autoridad de los Recursos Acuáticos de Panamá, 2008a) described mangrove clearing fees, where permit fees for commercial projects reached 150,000 balboas (at par with US$) per hectare, while illegal logging of mangroves was fined 300,000 balboas per hectare. In 2012, commercial permit fees were reduced to 10,000 balboas per hectare, and illegal logging fines were reduced to 40,000 balboas per hectare (Autoridad de los Recursos Acuáticos de Panamá, 2012). Reduced permit fees coincide with a suspension of the Panama Bay Wildlife Refuge, a protected site known for its rich biodiversity, migratory species, and importance for local fisheries (Romero Hernández, 2016). Suspension occurred for suspected reasons of urban expansion and "interest in facilitating mangrove conversion to commercial and residential developments" (Suman, 2014). Faced with the imminent threat of deforestation in Panama Bay, more than 50 NGOs and community groups from across the country joined and participated in advocacy work to reverse the ARAP's 2012 resolution (Romero Hernández, 2016). Protection status was reinstated in Panama Bay in 2015 (Asamblea Nacional, 2015a), despite illegal deforestation for luxury properties, golf courses, and shopping malls continuing to be reported in the protected area (Castellanos-Galindo et al., 2017). The Supreme Court of Panama reached analogous conclusions. Autoridad de los Recursos Acuáticos de Panamá's (2012) resolution was declared unconstitutional and was voided by the Supreme Court in 2016,

TABLE 2 | Summary of main national laws and policies on mangroves in panama since the 1941 constitution.

Law/Policy/Norm	Number and date of the norm	Institution	Key features	Current validity
Constitución Política de la República de Panamá	1941	Asamblea Nacional	• Before the 1941 Constitution, private property rights were recognized over coastal land, including mangroves. The Constitution declared that all coastal land was the property of the State.	A new Constitution was passed in 1972.
Constitución Política de la República de Panamá	1972	Asamblea Nacional	• The 1972 Constitution declared all coastal land and seas as public goods that are open to the public are free from privatization (article 258). The use of these public properties was granted *via* administrative concessions (Suman, 2002). • The State and all the inhabitants of the national territory must prevent pollution of the environment, maintain ecological balance, and avoid destroying ecosystems (Article 119). • The State guarantees that the use and exploitation of forests, lands, and waters are carried out rationally, to ensure their preservation, renewal, and permanence (article 120).	Amended in 1983, 1993, 1994, and 2004 without changing articles relating to mangroves.
Por la cual se aprueba la Convención Relativa a los Humedales de Importancia Internacional	Asamblea Legislativa, 1989	Asamblea Legislativa	• Panama approved the Ramsar Convention on Wetlands of International Importance and ratified it in 1992. Through the Convention, Panama commits to preserving the wetlands designated as Wetlands of International Importance: Bahía de Panamá, Golfo de Montijo, Damani-Guariviara, Punta Patiño, and San San Pond Sak. Many of these wetlands comprise mangrove forests. Panama also commits to improving the conservation and wise use of wetlands across time through collaboration with Ramsar offices.	Still in effect.
Por medio de la cual se dictan medidas para el uso y protección del manglar	Resolución J. D. 08-94	Instituto Nacional de Recursos Naturales Renovables (INRENARE)	• Mangroves are recognized as essential natural resources and their use becomes regulated. Logging by individuals is permitted, but a fee between 0.2 and 5 balboas is incurred. Logging for private purposes is also allowed, although an Environmental Impact Assessment and authorization are required first. Mangroves must be restored after logging.	Replaced by Autoridad de los Recursos Acuáticos de Panamá, 2008a (ARAP).
Ley General de Ambiente	Asamblea Legislativa, 1998	Asamblea Legislativa	• Mangroves are declared to have a high conservation priority because of their high biodiversity and productivity (article 95). • The Autoridad Nacional del Ambiente (ANAM) is created as the entity responsible for natural resources and the environment.	ANAM's responsibilities regarding mangroves are later transferred to ARAP.
Norma que crea la Autoridad de los Recursos Acuáticos de Panamá (ARAP)	Asamblea Legislativa, 2006	Asamblea Nacional	• ARAP is created and has the responsibility to manage coastal resources such as mangroves, in addition to establishing coastal management areas and ensuring compliance with the Ramsar Convention. ARAP also monitors water quality and all fisheries activities. • Mangroves are given a high conservation priority.	Management responsibilities over coastal resources are transferred to Ministerio de Ambiente in Law n° 8, 2015
Por la cual se establece el Comité Nacional de Humedales	Autoridad Nacional del Ambiente, 2007	Autoridad Nacional del Ambiente (ANAM)	• The National Committee on Wetlands is created as the inter-institutional organization bridging the Government and civil society to implement national wetland policies and support the Ramsar Convention. Its participating entities are enumerated, which include NGOs, universities, and governmental agencies.	Still in effect.
Que reconoce derechos posesorios y regula la titulación de tierras en zonas costeras e islas	Asamblea Nacional, 2009	Asamblea Nacional	• Land titling processes cannot include mangroves or protected areas. However, this law is not retroactive. This explains why there are currently private projects in mangroves (Tarté, 2013).	Still in effect.
Por medio del cual se establecen todas las Áreas de Humedales Marino-Costeros, particularmente los manglares de la República de Panamá, como zonas especiales de manejo marino-costero	Autoridad De Los Recursos Acuáticos De Panamá, 2008b	Autoridad de los Recursos Acuáticos de Panamá (ARAP)	• The responsibility to grant special permits for the sustainable use of the mangrove and collect fines in compensation for its damage is transferred to ARAP. • All mangrove areas are designated as marine-coastal management areas, where logging, use, commercialization, and deterioration are prohibited, with the exceptions of projects that receive approval according to other ARAP regulations.	Complemented by Autoridad de los Recursos Acuáticos de Panamá, 2008a (ARAP)

(Continued)

TABLE 2 | (Continued)

Law/Policy/Norm	Number and date of the norm	Institution	Key features	Current validity
Por la cual se aprueban algunas tasas y cobros por servicios que presta la Autoridad de los Recursos Acuáticos de Panamá	Autoridad de los Recursos Acuáticos de Panamá, 2008a	Autoridad de los Recursos Acuáticos de Panamá (ARAP)	• Permit fees for artisanal mangrove logging are established at 3 balboas per hectare per year. Commercial projects are subjected to fees of 150,000 balboas per hectare and illegal logging to a fine of 300,000 balboas per hectare.	Fines were reduced in Resolución J. D. 20, 2012.
Por la cual se modifica la Resolución J. D. n° 01 de 26 de Febrero de 2008, que aprobó algunas tasas y cobros por los servicios que presta la entidad	Autoridad de los Recursos Acuáticos de Panamá, 2012	Autoridad de los Recursos Acuáticos de Panamá (ARAP)	• Permit fees are reduced to 10,000 balboas per hectare, in addition to a requirement to reforest 2 hectares of mangroves per logged hectare. Fines for illegal logging are reduced to 40,000 balboas per hectare.	This resolution is declared void and illegal by the Supreme Court of Panama in 2016.
Que crea el Ministerio del Ambiente, modifica disposiciones de la Autoridad de los Recursos Acuáticos de Panamá	Asamblea Nacional, 2015b	Asamblea Nacional	• The Ministry of Environment is created, and all responsibilities for environmental protection, conservation, and management of coastal resources and transferred to this entity.	Still in effect.
Que establece la Política Nacional de Humedales del Estado en la República de Panamá	Ministerio de Ambiente, 2018	Ministerio de Ambiente (MiAmbiente)	• A new, more ambitious national wetlands policy is created. It is based on many principles: the precautionary principle, integrated ecosystem approach to wetland management, public participation, respect for cultural diversity, and adaptive management. This new approach aims to enhance the participation of civil society in wetland management and conserve wetlands to attain multiple Sustainable Development Goals. • The policy is enacted until 2050 and must be updated and evaluated every 5 years.	Still in effect.

citing that the resolution did not respect the State's will to guarantee a healthy environment and to avoid the destruction of ecosystems (Corte Surprema de Justicia, 2016).

While this Supreme Court decision points to an appreciation for the value of mangroves, mangrove policy frameworks in Panama remain nebulous. As raised by key informants involved in developing new coastal management policies, mangrove policies are confusing: "Two streams of regulations were kept moving forward, which today has brought us management problems deciding what standard should be applied" (participant from the Ministry of Environment). As shown in **Table 2**, regulations developed by different institutions are overlapping and remain in effect. For instance, ARAP no longer has policy jurisdiction over mangrove management, but some of their regulations are still applicable. Meanwhile, the Ministry of Environment developed recent policies (Asamblea Nacional, 2015b; Ministerio de Ambiente, 2018) with themes of sustainable use of mangrove resources by local communities, reduction of mangrove threats, and integrated management of wetlands, which competes with the other policy theme of commercial development on mangrove coasts. The internal contradictions in mangrove legal frameworks remain a major challenge in Panama. To resolve issues of inconsistent legal standards, the Ministry of Environment is developing an executive decree to unify current legislation on marine and coastal zones and to create appropriate rules that address the reality of mangrove loss and degradation (Ministerio de Ambiente, 2022). The proposed decree intends to create mangrove-specific protection measures and promote new standards of departmental coordination to effectively implement policies. However, special permits may be granted for projects related to tourism or broader public interest to be developed in mangrove forests upon approval of an Environmental Impact Assessment, which risks perpetuating existing patterns of mangrove clearing (Ministerio de Ambiente, 2022). Other measures include the creation of new mangrove restoration areas to counteract extensive habitat loss, and the implementation of Strategic Environmental Assessments to comprehensively analyze proposed projects in mangrove ecosystems. Fines for mangrove logging would return to the level described in ARAP's 2008 Resolution J. D. 1. The adoption of this decree, as well as its use in the context of development pressures remain to be seen.

Legislation relating to land tenure in mangrove forests also adds to the confusion. Legislation from the 1960s states that mangrove lands are public, except for mangroves already titled by private owners (Asamblea Nacional, 1962; Comisión Legislativa Permanente, 1964). The National Assembly passed a law in 2009 (Asamblea Nacional, 2009) permitting individuals who occupy land within the coastal zone to obtain a title from the government, although land titling processes cannot include mangroves or protected areas (Spalding et al., 2015). Due to the preemptive nature of this law, previously titled mangrove land remains private property and can continue to be developed (Tarté, 2013). These mechanisms facilitate the sale of coastal land for investment and create imbalances where land sales will mostly benefit elites and disempower local communities (Spalding et al., 2015). Questions have been raised on the willingness of public institutions to implement sustainable mangrove management across all zones (Suman, 2002), with issues of unclear land titles and non-compliance with laws reported by our key informants:

"*The Constitution states that mangroves and all wetlands belong to the government and not to private owners, but this does not apply to all people. It is a little bit contradictory. All the people that have owned land before the Constitution in the 1970s are not subjected to these regulations for the protection of mangroves. [. . .] This is a big issue*" (Government official).

"*Panama has signed the [Ramsar] Convention on Wetlands and was supposed to protect wetlands, but we have a big issue with private mangrove ownership. [. . .] Every time we make a law to protect mangroves, it does not continue because of this*" (Scientist).

Sectoral Responsibilities, Management, and Coordination

Mangroves have been under the jurisdiction of several government agencies. A wide range of agencies have had mangrove management responsibilities or have developed mangrove-related legislation, such as the Instituto Nacional de Recursos Naturales Renovables (INRENARE), Autoridad Nacional del Ambiente (ANAM), Autoridad de los Recursos Acuáticos de Panamá (ARAP), and the Ministerio de Ambiente (MiAmbiente) (see **Table 2**). Shifting institutional structure has led to an "institutional maze," where a lack of institutional memory has created high levels of confusion for the government and the public.

The Ministry of Environment (MiAmbiente) holds the central coordinating role in mangrove protection and inter-institutional collaboration. It exercises this authority employing the EIA process in which it must approve development projects across all sectors. However, current legislation fails to mention coordination between MiAmbiente and ARAP, who oversees the fisheries aspect of mangrove management. Some informants mentioned that coordination and communication are successful throughout the divisions of MiAmbiente that share responsibilities over mangrove forests: Dirección Forestal (Forestry Division), Dirección de Áreas Protegigas y Biodiversidad (Biodiversity and Protected Areas Division), and Dirección de Costas y Mares (Coasts and Ocean Division). Yet informants working for MiAmbiente did not mention coordinating with ARAP employees, and ARAP informants reported distrust with MiAmbiente's communication of information, in that little was shared with their institution:

"*It is hard to know how far their responsibility as an institution reaches and how far mine as an institution reaches, precisely because of this issue with the fishermen [who depend on mangroves]. We are the governing authority on the subject of fishing. How is it possible that they do not tell me anything?*" (ARAP informant).

Moreover, many of the most central actors in mangrove management and policy in Panama focus on the Ramsar Convention on Wetlands of International Importance, ratified in 1992. Panama purposefully created two institutions to advance the agenda of this convention, such as Ramsar-CREHO in 2003 (the Regional Ramsar Centre for the Western Hemisphere), and the National Committee on Wetlands (created through Resolución AG-0038 in 2007). Through concerted actions of academics, NGO leaders, government policymakers, as well as international cooperation with other Ramsar offices, these advisory bodies have multiple mandates: to manage wetlands, provide technical support to the National Government on wetland science and inventories, implement the National Policy on Wetlands, and promote outreach programs related to wetlands (Autoridad Nacional del Ambiente, 2007). Despite these mandates, many barriers reportedly slow their fulfillment. First, participants from the National Committee on Wetlands and Ramsar-CREHO outlined unclear responsibilities to develop and propose new regulations specifically designed for mangrove forests. Due to their ties to the government and their position as an advisory board, stakeholders in the National Committee are not in a suitable position to propose new legislation in the National Assembly and, instead, rely on non-governmental organizations to perform that task. Second, an important barrier was the lack of long-term financing. The Ministry of Environment's Division of Coasts and Oceans that also has a central role in wetland advisory boards, was reported to have a deficient operations budget and a lack of technical personnel. Monitoring of coastal habitats and patrolling are crucial operations amidst trends of deforestation, but they are also costly. While international funding is provided for sporadic initiatives, such as the "blue carbon" project that quantifies ecosystem services performed by coastal ecosystems (Ministerio de Ambiente, 2020; Ministerio de Ambiente, 2021), funding for monitoring and enforcement is intermittent. This situation can create significant power imbalances when facing corporate stakeholders, who may have a competing interest in mangrove management. More attention to the potential for public-private partnership models may be useful.

Research participants also described a general reluctance among government agencies to protect wetlands and a lack of political interest in that theme, aside from those governmental stakeholders directly involved in wetland advisory bodies. Some informants related this to private business interests, which allegedly interfere with State decisions:

"*We are talking about mangroves and suddenly someone comes with machinery. It is the tragedy of the commons, in the sense that generally mangroves are common lands of the State and many times there are the private interests of someone in particular who uses their economic or political influence to influence decisions, degrade the mangrove, and derive gain from those wetlands*" (Non-governmental organization).

"*Every time you do something about wetland protection, there is somebody trying to stop it*" (Government official).

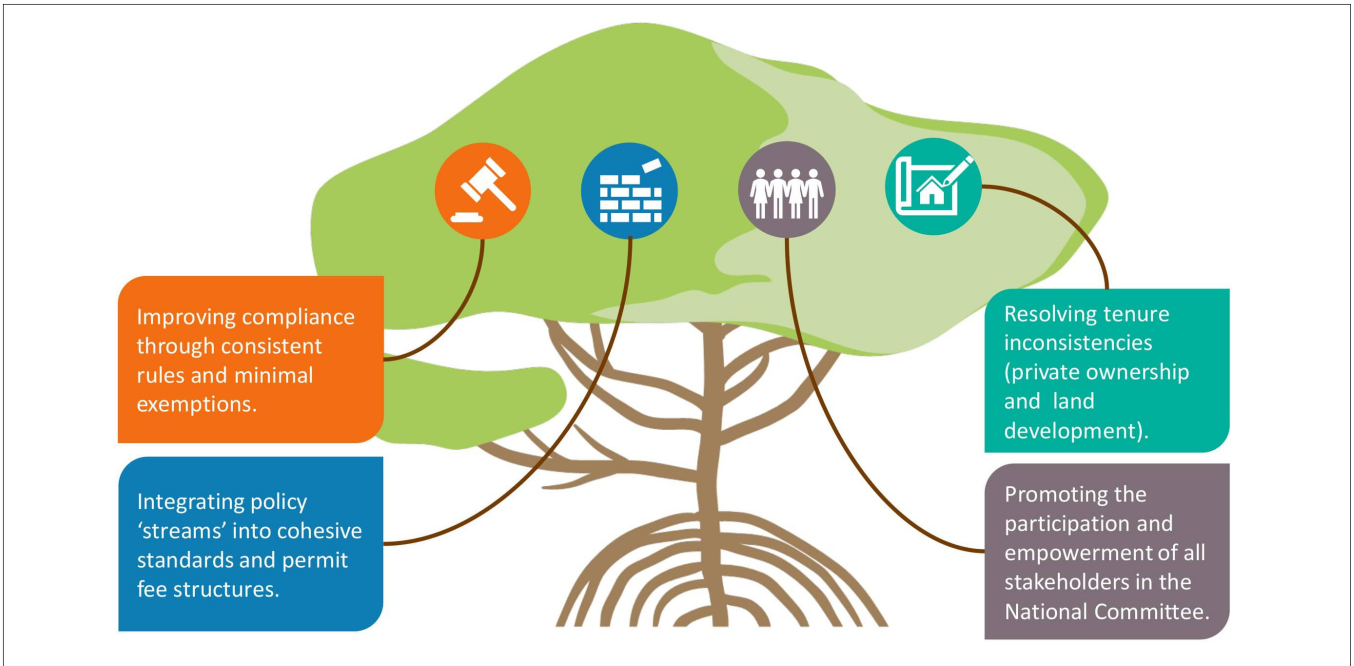

FIGURE 1 | Summary of policy barriers and pathways to foster sustainable mangrove management in Panama.

A summary of findings is presented in **Figure 1**.

DISCUSSION AND CONCLUSION

Unclear or Conflicting Policy Objectives and Antagonism With Private Sector Actors

Mangrove policies in Panama are characterized by their multiplicity and internal contradictions. Because of the variety of institutions that shaped policy, multiple "streams" of policy have been developed over time, even though they are not consistent with one another. Cross-sector links between policies also appear deficient (Suman, 2002). This "divergent evolution" of mangrove policies creates conflicting objectives and can eventually lead to implementation failures. According to policy implementation theory (Hudson et al., 2019), policy failure in Panama's mangroves has occurred in the first stage of policy implementation: policy design. Faulty policy design can stem from many causes: poor understanding of the problem; insufficient knowledge of the implementation context; unclear and even contradictory goals; and absence of political backing (Hudson et al., 2019). In Panama, many causes appear to be present. International NGOs and agencies, as well as many environmental scientists, have shifted their discourse in recent years, claiming that conservation and development goals need to converge (Ioris, 2014; Ministerio de Ambiente and the United Nations Development Programme [PNUD], 2018). Yet the discourse of wetlands as a conservation priority does not seem to appeal to elected officials, with economic development often prioritized over environmental conservation. While mangrove benefits in Panama are known and celebrated in key national policy documents (Kaufmann and Miró, 2012; Romero Hernández, 2016; Ministerio de Ambiente and the United Nations Development Programme [PNUD], 2018), shared understandings between stakeholders and by the public are lacking. Environmental conflicts opposing private sector actors and civil society are common in Panama's mangrove management context (Mejía, 2020) and are mirrored by conflicting directives from governmental agencies. The Panamanian government has evolved a regulatory system that fosters economic growth through foreign investment in coastal zones, at times at the expense of environmental preservation (Spalding, 2013; Thampy, 2014). Refusing to gain an advantage when economic opportunities arise can be viewed as "un-Panamanian," even when the alternative involves the protection of key ecosystems (Spalding, 2013; Thampy, 2014).

Numerous laws have been developed to address ecosystem preservation and establish a high conservation priority for mangroves, but they are not fully utilized. Examples of this include legal exceptions to mangrove protection for approved development projects, which effectively dominate the policy discourse. Furthermore, compliance with mangrove protection laws is challenging. Faced with the superior bargaining power of actors associated with coastal development, proposed development projects can be approved while established protected areas and international agreements, such as the Ramsar Convention, are ignored (Suman, 2014). This incapacity to deliver on commitments made under conventions, combined with the subsequent lack of trust in governmental institutions responsible for wetlands displayed by civil society, are signs of weak forest governance (Irland, 2008). Results from this study emphasize that mangrove mismanagement in Panama appears closely connected with competing agendas within government and pro-development politics that conflict with conservation policies, as argued by other authors (Suman, 2014; Castellanos-Galindo et al., 2017). These factors, combined with inadequate human and financial resources, mean further stages of policy implementation in Panama (tracking, implementation support, evaluation, policy review) have not yet been attained and could be further examined (Hudson et al., 2019).

In a systematic literature review on sustainability policy failure, Howes et al. (2017) found that recurring causes of implementation failure include the preference for economic outcomes over environmental ones, concern with market failure, and the lack of market instruments to address environmental issues. These findings apply to the context of mangrove management in Panama. To move beyond the expected environmental versus development trade-off, several studies have highlighted opportunities for greater private sector engagement in mangrove management. Private sector participation could, for example, strengthen the idea that conservation and development are not necessarily antagonistic and can foster more cooperative relationships between stakeholders (Nickerson, 1999; Friess et al., 2016). Private-sector approaches to mangrove management include traditional unilateral donors, corporate social responsibility initiatives, and market-based ecosystem service instruments (Friess et al., 2016). Of particular interest to Panama is the payment for ecosystem services (PES) tool, which can "address overlapping or conflicting policy objectives by [. . .] allowing stakeholders from community to national levels to coalesce around a clear PES objective" (Friess et al., 2016, p. 941). Due to heavy investments in Central America for "blue carbon" projects that require ecosystem service quantification, Panama appears well placed to engage in PES with private sector actors (Ministerio de Ambiente, 2021). This approach could also help generate much-needed funds for mangrove restoration and conservation.

Alternatives to private sector engagement could focus on stricter legal frameworks, an approach favored by many mangrove-bearing countries (Slobodian and Badoz, 2019). Environmental law "slippage," whereby compliance with laws is deficient and regulators fail to act on transgressions (Farber, 1999), was observed in our study and other mangrove management studies in Panama (Suman, 2014; Castellanos-Galindo et al., 2017). Lessons may be offered by Costa Rica and Chile's examples, having established an Environmental Administrative Tribunal as a mechanism enforcing environmental regulations, imposing sanctions, and applying interim protection measures after legal transgressions of different stakeholders, including land-use change in urban areas (Slobodian and Badoz, 2019). Similar tribunals adjudicating for sustainable mangrove management cases are also present in Kenya and India (Slobodian and Badoz, 2019). Nevertheless, in contexts where the government is considered complicit with transgressions, stronger enforcement measures and focus on compliance may be misguided and ineffective. Greater emphasis could be put on devolving more power to multi-party institutions like the National Committee on Wetlands, who are already dedicated to aligning policies with international discourses on habitat conservation, ecosystem services, and nature-positive cities. By including new actors in this committee, such as community representatives, such institutions could be better positioned to promote a more sustainable—and participatory—approach to wetland management.

Collaborative Management

Policy implementation failure can also be related to a lack of continuous collaboration between the multiple stakeholders at the political, policymaking, managerial, and administrative levels, as well as the lack of engagement of end-users and local communities (Hudson et al., 2019). This connects with Panama's history of agency overlap, confusing institutional landscape, and multiple policy "streams" (Spalding et al., 2015). Mangroves have been governed by at least twenty laws and policies, overseen by six different institutions (Tarté, 2013). These complex governance environments are common in mangrove forests but are known to impede coherent policy formation and leave agencies with conflicting aims and responsibilities (Friess et al., 2016). Due to recent policy updates, Panama has established central coordinating agencies that oversee mangrove management: The Ministry of Environment, in addition to the Ministry of Housing and Land Use Planning (MIVIOT) who is responsible for municipal land use plans. However, coordination and regular communication beyond the Ministry's divisions and across agencies were still reported to be challenging. Collaborative management strategies could help to address some of the issues identified, while also opening forest management discussions to other stakeholders.

Most importantly, more attention may be devoted to the influence of multiple parties in mangrove management with an emphasis on identifying which actors are—and are not—participating (Safford, 2012). Multi-stakeholder partnerships could include actors within universities, NGOs, coastal communities, and the private sector. Industry and business sector organizations have substantial influence over mangrove management, as seen in the case of Panama Bay (Suman, 2014; Castellanos-Galindo et al., 2017), but they tend to be peripheral players in multi-party management efforts, such as the National Committee on Wetlands. Political lobbying combined with an absence from multi-party processes have possibly impeded collaboration on mangrove management (Safford, 2012). As argued by Safford (2012), wetland managers could better acknowledge the political nature of management activities and illustrate to politically engaged actors that multi-party planning does not undermine their interest. Yet, when reuniting actors with vast power asymmetries, collaborative and equitable outcomes can be hindered, especially since lasting antagonism between land developers and coastal communities has led to environmental conflicts in the past (Mejía, 2020). Bringing these groups together and applying conflict resolution and mediation techniques have the potential to diffuse tensions and build a foundation for greater consensus (Safford, 2012).

Currently, local communities are also peripheral actors in Panama's mangrove management. Mangrove-dependent communities are closely intermeshed with ecosystem-level outcomes for reasons of resource use and poverty alleviation. The inclusion of local communities is likely to be particularly important to avoid restricting community use of mangroves (Dev Roy, 2012; DasGupta and Shaw, 2017; Félix and Hurtado, 2019), as well as to address underlying issues of illegal logging and poaching, which are often connected to unresolved property rights (Clarke et al., 1993; Amacher, 2009). The needs of local communities in Panama's coastal management have been given scarce policy attention, as shown by evidence of unfair property rights and access to coastal zones (Spalding et al., 2015), deforestation of habitats that support artisanal fisheries (Suman, 2014), and absent community representation in management boards such as the National Committee on Wetlands. Collaborative management has the potential to reorient conversations about mangroves back to its primary users amidst trends of privatization of coastal land (Spalding et al., 2015). Recent policy developments such as the National Policy on Wetlands identify objectives of integrated coastal zone management (ICZM) and participatory approaches, yet mechanisms to devolve power to communities and move beyond consultation are unspecified (Ministerio de Ambiente and the United Nations Development Programme [PNUD], 2018). To ensure participation is effective and inclusive, participatory management in mangroves requires rigorous incentive design (DasGupta and Shaw, 2017). This is especially relevant due to historical inclinations of "top-down" forest management, strict control, and patrol of forests, which may create path-dependency and strong inertia toward institutional change. Without clearer roles for local communities in existing institutions, Panama risks further antagonizing its mangrove users.

Further research on the strategies of fisher groups, local resource users, and NGOs when facing power imbalances with private sector actors who interact frequently with natural resource management professionals and apply coercive pressure would be beneficial. In multi-actor management boards such as the National Committee on Wetlands, research to clarify the relationships between all participating actors, level of internal consensus, effective coordination strategies, shared recognition for the utility of collaborative inputs, consistent participation, and power differentials would also be valuable. Contemporary insights to these collaborative processes could help to clarify avenues for a more sustainable approaches to mangrove management, whereby multi-actor committees and civil society play a more active role.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author/s.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by McGill University Research Ethics Board (REB File #: 19-11-046). The patients/participants provided their written informed consent to participate in this study.

AUTHOR CONTRIBUTIONS

SC-F: conceptualization, data curation, formal analysis, investigation, methodology, project administration, and roles/writing—original draft. GH: conceptualization, methodology, project administration, resources, supervision, writing—review, and editing. SH-M: conceptualization, supervision, and editing. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/ffgc.2022.818722/full#supplementary-material

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Sedimentation as a Support Ecosystem Service in Different Ecological Types of Mangroves

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Mangrove vegetation is strongly dependent on the climate, the physicochemical variables of the sediment, and the hydrological dynamics. These drivers regulate the distribution of different mangrove ecotypes and their ecosystem services, so the net sediment accumulation rates in different mangrove ecotypes in Celestun Lagoon, a karstic zone in the NW Yucatan Peninsula, SE Mexico, were estimated. The measurements considering mangrove ecotypes and their spatial variability concerning the lagoon's salinity gradient (inner, middle, and outer lagoon zones) in three climate seasons (dry, rain, and "nortes") were realized. We registered the structural variables of the forest, interstitial water physicochemical characteristics, and sediment variables that could influence the net sediment deposition. Fringe mangroves are exposed to low hydrodynamism and show the highest sedimentation rate (3.37 ± 0.49 kg m^{-2} $year^{-1}$) compared to basin (1.68 ± 0.22 kg m^{-2} $year^{-1}$), dwarf (1.27 ± 0.27 kg m^{-2} $year^{-1}$), and "peten" (0.52 ± 0.12 kg m^{-2} $year^{-1}$) mangroves. The highest sedimentation rate was recorded in the rainy season (0.24 ± 0.08 kg m^{-2} $month^{-1}$), while spatially, the highest value was registered in the outer zone (0.44 ± 0.09 kg m^{-2} $month^{-1}$). If the extension of each mangrove ecotype is considered, dwarf mangroves have the highest annual sediment accumulation (1,465 t $year^{-1}$ in 14,706 ha). The structural, physicochemical, and sediment variables of the sites by mangrove ecotype show that dwarf mangroves represent a distinct group from those formed by fringe, basin, and peten mangroves. However, the sedimentation is high in fringe mangroves at the front of the lagoon and diminishes inland where peten mangroves exist. The differences are given by tree density, but salinity, as a proxy variable of the freshwater influence, significantly influences the sedimentation rate. These results indicate that mangroves in karstic environments can have critical roles in confronting climate change, considering water and sediment flows are the basis of sediment accumulation. According to their hydrogeomorphological drivers, conserving, managing, and restoring the mosaic of mangrove ecotypes improves ecosystem services, including mitigation and adaptation to climate change.

Keywords: mangrove types, sedimentation rate, ecosystem services (ES), conservation, mitigation and adaptation, karstic area

INTRODUCTION

Mangroves grow in the sea–land confluence zone of tropical and subtropical regions. This ecosystem shows variability in its vegetation characteristics and adaptations like prop roots and succulent leaves (Naskar and Palit, 2015). Different forest types can be observed according to their vegetation structure, growing in diverse areas with hydrological, physicochemical, and sediment characteristics (Middelburg et al., 1996). The sediment accumulation in mangroves shows patterns (Adame et al., 2010) at spatial and temporal scales related to hydrodynamics (e.g., floods, water flows, precipitation, tides, surges, and storms), which, in turn, control organic and inorganic sediment supplies (Woodroffe et al., 2016), modulated by terrain slopes, topography, and geomorphological features (Twilley and Rivera-Monroy, 2009; Cannon et al., 2020; MacKenzie et al., 2021).

The interactions between these characteristics generate many ecosystem services (Getzner and Islam, 2020). Some of them are related to reducing mangroves and adjacent habitats' vulnerability to climate change impacts, including supporting ecosystem services originating from key ecological processes such as soil formation, nutrient cycling, and primary productivity. These processes are essential for other mangrove ecosystem services, such as wood and food provision (Mitsch et al., 2015).

The sediment capture is one of these critical ecological processes that contribute to a supporting ecosystem service, as the sedimentation in coastal ecosystems is associated with the regulation of tidal flow speed (Kobashi and Mazda, 2005) and the concentrations of suspended sediment and organic matter in the water flowing through mangroves (Kobashi and Mazda, 2005; Adame et al., 2015; Friess and McKee, 2021). There is a direct relationship between features such as tree density and dead trunks and water flow, thus determining sedimentation patterns in mangroves (Mazda et al., 1997; Montgomery et al., 2019).

In response to mesoscale processes related to hydrodynamics and sediment supply, the mangrove trees develop different associations and structures of vegetation that originate diverse mangrove ecological types (mangrove ecotypes): fringe, basin, dwarf, hammock, overwash, and riverine (Lugo and Snedaker, 1974), which show different functioning that may provide ecosystem services at different intensities (Agardy and Alder, 2005).

The mangrove capacity to capture sediments is reflected in the relatively high carbon accumulation observed (5 to 1,722 g C m^{-2} $year^{-1}$; MacKenzie et al., 2021). This carbon accumulation in mangrove sediments contributes to other ecosystem services that mangroves provide, such as greenhouse gas regulation, removal of atmospheric CO_2, coastal zone protection against sea storms, and sea-level rise through the vertical elevation of soil (Alongi, 2008; McKee, 2011). The accumulation of carbon in the sediment favors an increase in the soil level. This accumulation must be greater than the decomposition–respiration processes for the vertical balance of the soil to be positive. If the capture of C in the soil is not sufficient to overcome subsidence, the site sinks and is vulnerable to sea-level rise, causing flooding and putting the stability of the coastal zone at risk (Spalding et al., 2014). Although the rates of carbon sequestration in sediments are relatively high in the process of vertical soil accumulation, there are other factors involved, such as accretion, compaction, water flows, and the composition and formation of soil by living organisms, as well as deeper processes at the regional level (Lynch et al., 2015), that must also be considered.

The source of the sediments deposited in a determined zone is an essential factor influencing the sediment accumulation rate in different mangrove ecotypes. According to the geomorphological features of mangroves, their sediments can have different origins, so the mineral composition and the size of the particles that settle can be determined. The size of the particles is crucial because it is involved in the dispersion: small particles are usually transported greater distances and settle when the energy decreases; furthermore, the size of the sediment particles is related to the absorption of organic material and therefore to the carbon content at the site.

The use of stable isotopes, mainly carbon and nitrogen, is a valuable tool to determine the transference routes of materials (Adame and Fry, 2016), which is essential to identify the origin and transport routes of organic and inorganic material in sediments. This information allows us to deduce the role of an ecosystem as a sink, transformer, or source of organic material. In the places with relatively low hydrological dynamics (low energy level, small tidal ranges, no surface currents), mangroves are transformers and carbon sinks, which is reflected in high capture rates in the soil. The drivers that influence the accumulation and retention of sediments have been studied little (MacKenzie et al., 2021), but they respond to the influence of the local characteristics of the area. Knowing the sediment sources, such as where they come from, how the sediments support the processes, and functions of the mangrove forest is an essential tool for decision-making on management, considering the local dynamics.

The knowledge of the functions and processes of mangroves is the starting point for adequate management of the ecosystem, including decisions related to conservation and restoration based on these features that result in an essential tool for adaptation and mitigation to climate change. Knowing the factors associated with the processes; for example, what characteristic is responsible for the role of a sink, source, or transformer that a certain mangrove plays, as well as the source of sediments and carbon, helps to spatially and temporally delimit the implementation of actions that achieve conservation and restoration objectives.

However, mangroves are differentially vulnerable to climate change impacts according to variables related to hydrodynamics (Cinco-Castro and Herrera-Silveira, 2020). This hydrodynamic in the coastal zone is modified according to predictions on changes in precipitation patterns and sea-level rise (IPCC, 2021) because the water dynamics at a site depend on what happens in the basin and how much water reaches the coastal area. The predictions related to more or less precipitation then influence the hydroperiod, salinity, redox potential, productivity, and decomposition processes of organic matter in mangroves, which translates into different sedimentation rates.

As sedimentation dynamics in mangroves contribute to conserving these ecosystems for a long time, maintaining and improving their hydrologic and topographic characteristics should be part of management policies (Chow, 2018). These characteristics guarantee ecosystem services at local, regional, and global scales. Based on the above, the objectives of this study were to determine differences in net sedimentation rates in different mangrove ecotypes distributed in a spatial gradient. The gradient is due to salinity in the lagoon and functions as the main stressor in mangrove ecosystems. The temporal variability takes into account distinct climatic seasons. Also, variables that influence sediment capture changes are identified, considering the sediment origin and composition.

STUDY AREA

The study area is in the mangrove forest in the north of Ria Celestun Biosphere Reserve (RCBR) in the NW Yucatan Peninsula (SE Mexico, **Figure 1**). The Yucatan Peninsula is geomorphologically formed by Tertiary limestone with high infiltration potential, which causes the absence of surface water but high groundwater flows (Batllori-Sampedro, 1995). The soils are shallow limestone, and topographically, the soil surface slope is less than 1% (SEMARNAT, 2000). These settings create an aquifer with a hydraulic gradient of 7–10 mm km^{-1} (Rey, 2012). The climate is predominantly dry, with an average annual precipitation of 759 mm but high interannual variability (395–1,239 mm year^{-1}). However, there is a marked seasonal variation, with a dry period from March to May, heavy tropical rainfalls from June to October, and soft rains from November to February. This last season, called the "Nortes" season, represents a meteorological phenomenon that usually appears with rains, wind, and surges that change the hydrologic dynamics in the coastal zone.

Ria Celestun is a 22-km-long coastal lagoon with an average width of 1.25 km and an average depth of 1.2 m (Acosta-Lugo et al., 2010). This lagoon has a marked spatial salinity gradient that varies with the seasonal meteorological characteristics mentioned above (Herrera-Silveira, 1994) and the sea–lagoon flows, which establish minimum and maximum water residence times between 7 and 63 days, respectively, at the outer and inner parts of the lagoon (Herrera-Silveira and Comin, 1995). The groundwater springs are freshwater inputs with a maximum flow of 7 m^3 s^{-1} during the rainy season and a minimum of 1.2 m^3 s^{-1} in the dry season (SEMARNAT, 2000). The tides are 0.76 m in breadth (Torres-Mota et al., 2014) and they have a propagation efficiency in the aquifer of 39% (Rey, 2012).

Celestun has a great diversity of submerged aquatic vegetation, coastal dunes, mangroves, water bodies, floodplains, and lowland rainforests. The mangrove forests in the reserve polygon cover ∼45,000 ha (CONABIO, 2021), and based on vegetation structural characteristics, the following mangrove ecotypes could be observed: fringe, basin, dwarf, and peten. On the Yucatan Peninsula, peten refers to a type of mangrove growing like a vegetation island associated with springs or "ojos de agua" that provide freshwater. These water inputs are characterized by low salinity and high nutrient concentrations, mainly nitrates, favoring less stress and resources available for tree growth. The Peten term is similar to the "hammock" mangrove ecotype described by Lugo and Snedaker (1974) in Florida.

Within the limits of the RCBR, the use of natural resources is made both for self-consumption and for commercial purposes through fishing, salt extraction, and ecotourism projects (SEMARNAT, 2000; Ramsar, 2004). Among the identified threats are mangrove and submerged aquatic vegetation losses, lagoon siltation, and eutrophication; however, altogether, mangroves in Celestun are considered in good condition from an ecological conservation point of view (Herrera-Silveira and Morales-Ojeda, 2010).

METHODOLOGY

Mangrove Ecological Types

A satellite image analysis was carried out to obtain the mangrove extent in the study area. The mage from 1 March, 2020, from 21–45 scene (path and row), referred to Worldwide Reference System, from Landsat 8 OLI with 30 m resolution were downloaded from USGS Earth Explorer. During the preprocessing, an atmospheric correction was applied using the dark object subtraction (DOS) method, one of the most effective atmospheric correction methods (Wicaksono and Hafizt, 2018). The DOS directly transforms the digital values to reflectance, assuming reflectance from dark objects includes the atmospheric scattering (see **Supplementary Annex A**).

A preliminary visual analysis was carried out using a "false color" composition [Near-Infrared (NIR), red, and green]. For mangroves, five classes were observed according to their reflectance (fringe, basin, dwarf, disperse dwarf, and peten). Additionally, dune vegetation, without vegetation zones, and saline soil areas were identified. The polygonal "training sites" were defined using false-color composition and *in situ* verification for each class. A supervised classification was carried out using the maximum likelihood algorithm considering bands 2, 3, 4, and 5. The Kappa index (range 0 to 1) and error matrix were obtained to indicate classification accuracy based on classification and reference data differences.

For the monitoring of filed, 16 permanent plots (100 m^2 each) representing different mangrove ecotypes, fringe (5), basin (7), dwarf (2), and peten (2), were selected to be studied according to the zone's knowledge. These plots were distributed in a spatial gradient given by salinity in three zones of the east shore of Celestun Lagoon: (a) the inner zone, characterized by freshwater inputs due to numerous springs (North); (b) the middle zone, represents the central point of the lagoon and is a mixing area where freshwater from the inner zone and marine water from outer zone converge; (c) the outer zone (South), located in the coastal lagoon inlet showing the marine influence and generally high salinity to the rest of the lagoon (**Figure 1**). The differences in permanent plots number among mangrove types are due to fringe and basin mangroves that exhibit a significant variability around the lagoon gradient compared with dwarf and peten mangroves that are usually more homogeneous. The forest structure vegetation was recorded in each plot once, and environmental variables (physical and chemical characterization) were sampled in all the plots in three seasons: (a) dry (March to May), (b) rains (June to October), and (c) "nortes" (November to February).

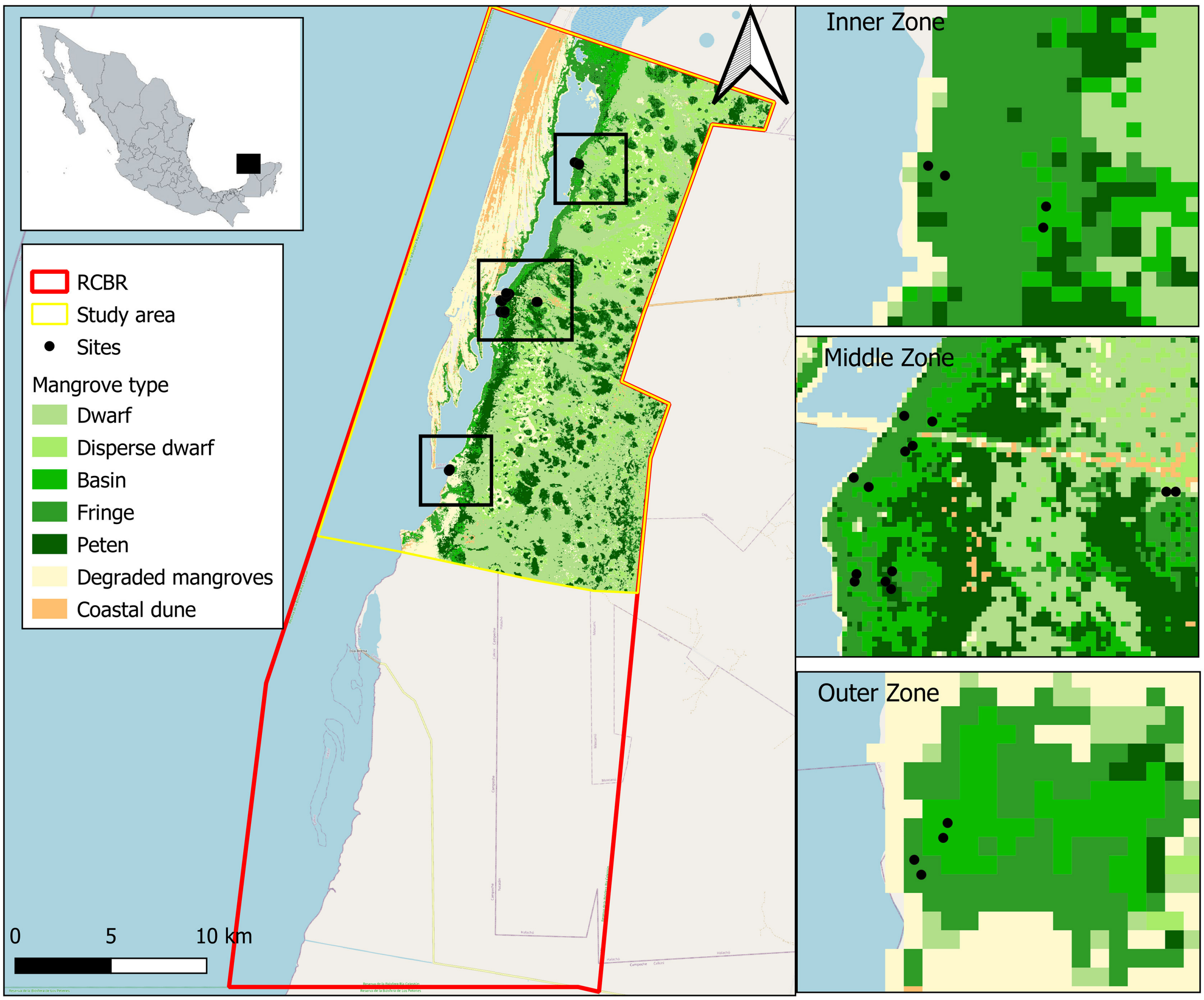

FIGURE 1 | Study sites in Celestun (Yucatan, México). Location by lagoon zones (distance among Inner, Middle, and Outer zones is 8 km on average) and by mangrove ecological type (distance among fringe, basin, dwarf and peten mangroves varies from 300 to 1,500 m).

Structure Vegetation Variables

To determine the forest vegetation structure in each mangrove ecotype, species, height (H), and the diameter at breast height (DBH) in all trees with diameters greater than 2.5 cm were recorded in each plot according to the methods described by Rodríguez-Zúñiga et al. (2018). From these data, basal area (BA, m^2 ha^{-1}) and density (D, ind ha^{-1}) were calculated, including seedling density at each site (ind m^{-2}). Seedling density is an indicator of natural regeneration related to mangroves' resilience capabilities (Ellison, 2012).

Sedimentation Rate

Five sediment traps were located at the soil surface level in each permanent plot to measure the sedimentation rate in dry weight per area per time. Traps were created using filter paper on 8.2 cm diameter Petri boxes and metal mesh as protection. Wire fasteners were used to fix traps in soil (**Supplementary Figure A**).

The traps were exposed in the field for 30 days in each climate season in 2019: a) dry (from 30 April to 30 May), b) rains (from 23 August to 22 September), and c) "nortes" (5 November to 5 December). Later, the traps were collected and carried to the laboratory. The materials such as leaves, trunks, and unidentified elements were separated, and sediments were dried using an oven at 70°C for 72 h to obtain the dry weight. The sedimentation rates were calculated using dry weight, filter area, and exposure days, according to the following equation:

$$\textbf{Sedimentation rate}\left(\textbf{g cm}^{-2}\textbf{day}^{-1}\right) = \frac{\textbf{filter weight } \textbf{t}_1\ (\textbf{g}) - \textbf{filter weight } \textbf{t}_0\ (\textbf{g})}{\textbf{filter area}\left(\textbf{cm}^2\right) * \textbf{number of days}}$$

The sedimentation rate was annually extrapolated by mangrove ecotype as kg m^{-2} $year^{-1}$, considering the seasonal rates and the extension of the respective mangrove ecotypes in the RCBR polygon.

Environmental Variables

The environmental variables influence mangrove structure and provide information about the processes occurring in the ecosystem. These processes can be related to the sedimentation rate so that, in each plot, three samples of interstitial water at 30 cm depth were collected using a syringe and perforated acrylic tubes. The salinity was recorded using an Atago refractometer. The temperature, pH, and redox potential were recorded using an Ultrameter II™ 6PFCE device. Furthermore, in each plot, three aboveground flood levels were recorded, measuring 3 times the water height from the soil surface to the water surface using a 1 m ruler at the same sites as sediment, and environmental variables were recorded.

Sediment Source and Characteristics

A surface sediment sample (0–15 cm deep) was collected using a metal nucleator 5.25 cm in diameter in each permanent plot. The samples were characterized and dried at 70°C for 72 h. The measures bulk density (BD), organic matter content, total phosphorous, nitrogen, and carbon content in sediments were measured to determine sediment variables and register how they change spatially and temporally. The BD was obtained from dry weight and sample volume quotient. The organic matter content was determined using the weight difference after sediment calcination at 550°C for 4 h (Chen and Twilley, 1999). The phosphorous content was determined using Strickland and Parsons's (1972) methodology. In sediments, the total nitrogen and carbon (%) were measured in an elemental autoanalyzer model Flash-EA-1112 using 20–30 mg of previously ground and homogenized samples. The organic carbon content was determined from the difference between the total and inorganic carbon determined after the ignition method (Heiri et al., 2001).

Cores of 5.25 cm in diameter and 5-cm deep surface sediment were taken for organic matter isotopic composition analysis to establish the sediment origin in each mangrove type. The isotopic signatures ($\delta^{13}C$, $\delta^{15}N$) were determined after grounding the samples with an agate mortar and were analyzed with an isotope ratio mass spectrometer and an elemental autoanalyzer. The calibration was based on acetanilide. The procedure was performed in duplicate, with 0.2% accuracy in both cases. $\delta^{13}C$ and $\delta^{15}N$ were calculated using the ^{13}C:^{12}C and ^{15}N:^{14}N relations of the sample (Rx) and standard (Rs) according to the following formula:

$$\delta\,[\text{‰}] = \left(\frac{\mathbf{R_s}}{\mathbf{R_x}} - \mathbf{1}\right) * \mathbf{1000}$$

A mixed model diagram was used to determine the possible sediment sources, considering $\delta^{13}C$ and N:C, and to compare our data in sediments with mangrove leaf values, senescent mangrove leaves, mangrove sediments, C3 plants, and mangrove-associated herbs (*Batis maritima* and *Salicornia virginica*). A rearranging isotopic mixing equation was applied (Shultz and Calder, 1976) to determine the autochthonous carbon fraction, comparing terrestrial source (mangrove leaves) and marine (phytoplankton) values with values in our sediment samples:

$$\mathbf{F_t} = \frac{\partial \mathbf{C^{13}_{sediment}} - \partial \mathbf{C^{13}_{marine}}}{\partial \mathbf{C^{13}_{terrestrial}} - \partial \mathbf{C^{13}_{marine}}}$$

Data Analysis

A modified Shapiro–Wilks normality test determined the sedimentation rate differences between mangrove types, seasons, and lagoon zones. A multifactorial analysis of variance (ANOVA) was run to fulfill this assumption. The same procedure was carried out for structural, interstitial water physicochemical, and sediment variables, including isotopic signatures.

The Similarity Percentages analysis (SIMPER) was carried out to determine the dissimilarity between mangrove ecotypes. The Pearson correlation coefficients were considered to identify correlated variables. A linear discriminate analysis (LDA) was carried out to determine the relationships of structural, physical–chemical, and the sediment characteristics with sedimentation rate, considering only variables that do not correlate between them. All data analyses were carried out using R 3.6.3 and InfoStat 2019.

RESULTS

Mangrove Ecotypes

The study area is a polygon in the north of the RCBR whose terrestrial and marine environments extend 42,739 ha. The image classification accuracy is 91.2% and 0.9058 for the Kappa coefficient. All classes were classified with accuracy more than 80% (**Supplementary Figure B**). Based on this, mangroves cover 26,133 ha, and the spatial distribution is related to the zone's characteristics, such as hydrology and topography. Fringe mangroves cover 2,735 ha bordering the lagoon and in the coastal zones where mangroves are strongly influenced by the ebb and flow of the tide. Basin mangroves have an extension of 811 ha and are located behind the fringe mangroves, in zones where the topographic profile is usually lower, favoring water accumulation during the ebb and increasing water and soils salinities due to high evaporation. The peten mangroves are identified as the highest vegetation (>15 m) due to freshwater discharges to which they are associated. This mangrove type has an extension of 4,011 ha. Finally, dwarf mangroves are smaller trees (see Results section) because they grow in areas with nutrient limitations. Despite this, dwarf mangroves cover 14,706 ha. However, in Celestun, dwarf mangroves can be associated with other types of wetlands inland (**Figure 1**).

Vegetation Structure

Mangroves in the Celestun area have structural characteristics of the vegetation that define the ecotype. The data show that the peten mangroves ($n = 55$) have higher DBH and height (29.8 ± 2.2 cm and 14.8 ± 2.0 m, respectively) than fringe ($n = 150$, 20.2 ± 1.4 cm, and 11.7 ± 1.3 m) and basin mangroves ($n = 327$, 16.1 ± 1.2 cm, and 11.1 ± 1.0 m, respectively). The dwarf mangroves ($n = 119$) are, on average, the smallest (1.8 ± 2.4 cm, 1.67 ± 2.2 m), but they show high tree density (26,444 ± 1,393 ind

ha^{-1}), which makes them significantly different from the other types ($p < 0.05$). Seedling densities vary from 6 to 12 ind m^{-2}; however, the mangrove types are not significantly different when we consider this variable.

According to the lagoon zones, DBH and height were significantly higher in the inner zone ($n = 129$, 23.80 ± 1.4 cm, and 18.26 ± 0.5 m, respectively) than in the middle ($n = 369$, DBH: 15.66 ± 1.3 cm, H: 8.59 ± 0.6 m) and outer zones ($n = 153$, DBH: 13.63 ± 1.4 cm, H: 7.09 ± 0.6 m) ($p < 0.0001$). However, the density and the basal area were higher in the middle zone (D: 5,481 ± 929 ind ha^{-1}, BA: 73.91 ± 5.8 m^2 ha^{-1}) than in the other zones of the lagoon ($p < 0.05$). The seedling density is higher in the middle zone of the lagoon (12 ind m^{-2}) than in the outer (9 ind m^{-2}) and in the inner zones (7 ind m^{-2}).

Sedimentation Rate

The highest sedimentation rate was registered in fringe mangroves (3.37 ± 0.49 kg m^{-2} $year^{-1}$), followed by basin and dwarf mangroves (1.68 ± 0.22 kg m^{-2} $year^{-1}$ and 1.27 ± 0.27 kg m^{-2} $year^{-1}$, respectively). The lowest value was observed in peten mangroves (0.52 ± 0.12 kg m^{-2} $year^{-1}$). Therefore, a decreasing pattern in sedimentation rate is related to the spatial distribution of mangrove types, from fringe mangroves at the lagoon edge to peten mangroves located further inland in the study area. In general, the mean sedimentation rate for the mangroves in Celestun is 1.71 ± 0.60 kg m^{-2} $year^{-1}$.

In fringe and basin mangroves, the highest sedimentation rate was registered during the rainy season (0.24 ± 0.08 kg m^{-2} $month^{-1}$); it decreased during the nortes period (0.19 ± 0.03 kg m^{-2} $month^{-1}$), and the lowest value was registered during the dry season (0.04 ± 0.007 kg m^{-2} $month^{-1}$) (**Table 1**). Then, the sedimentation rate shows a decreasing trend as precipitation decreases in these two mangrove ecotypes. However, this pattern is not observed in dwarf and peten mangroves, with the highest sedimentation rates recorded during the nortes season.

According to the location of mangroves along the coastal lagoon, mangroves in the inner zone have a sedimentation rate of 0.05 ± 0.007 kg m^{-2} $month^{-1}$, followed by middle zone mangroves with 0.06±0.01 kg m^{-2} $month^{-1}$. The highest value was registered in the outer zone (0.44 ± 0.09 kg m^{-2} $month^{-1}$).

The statistical analysis shows that the most significant effect on the sedimentation rate is given by the position concerning the lagoon ($F_{2,2\ 3} = 14.05$, $p = 0.0001$), and to a lesser extent, by the season ($F_{2,23} = 4.01$, $p = 0.03$) (**Figure 2**). There exists little interaction between these factors ($F_{4,23} = 3.04$, $p = 0.04$) that influences the sedimentation rate in Celestun mangroves.

The dwarf mangroves have a more extension (14,706 ha) than other mangrove ecotypes in the study area, thus capturing approximately 1,465 t $year^{-1}$ of sediment under a conservative approach. However, the fringe mangroves have an extension of 2,735 ha and show the second-highest total sedimentation (923 t $year^{-1}$), followed in magnitude by peten mangroves (211 t $year^{-1}$ in 4,011 ha) and basin mangroves, which have 811 ha in extension with a total sedimentation rate of 136 t $year^{-1}$. The total sediment capture in the mangroves north of the RCBR is estimated at 3,483 t $year^{-1}$.

TABLE 1 | Sedimentation rate (kg m^{-2} $month^{-1}$) in four mangroves ecotypes during three climate seasons.

Mangrove type/Season	Sedimentation rate (kg m^{-2} $month^{-1}$)				
	n	Mean	SE	Min	Max
Fringe	**57**	**0.26**	**0.08**	**0.003**	**4.60**
Rain	14	0.49	0.31	0.020	4.60
"Nortes"	24	0.29	0.07	0.026	1.57
Dry	19	0.07	0.02	0.003	0.27
Basin	**99**	**0.14**	**0.02**	**0.00002**	**1.13**
Rain	32	0.24	0.06	0.00002	1.13
"Nortes"	34	0.14	0.04	0.016	1.07
Dry	33	0.04	0.01	0.031	0.23
Dwarf	**23**	**0.08**	**0.04**	**0.002**	**0.92**
Rain	10	0.08	0.02	0.019	0.22
"Nortes"	5	0.23	0.15	0.010	0.92
Dry	8	0.00	0.0001	0.002	0.01
Peten	**27**	**0.05**	**0.02**	**0.000002**	**0.45**
Rain	7	0.02	0.01	0.002	0.04
"Nortes"	10	0.10	0.04	0.016	0.45
Dry	10	0.01	0.0003	0.000002	0.03
MEAN	**206**	**0.15**	**0.03**	**0.002**	**4.60**
Rain	63	0.24	0.08	0.002	4.60
"Nortes"	73	0.19	0.03	0.010	1.57
Dry	70	0.04	0.01	0.031	0.27

Values are samples (n), mean, standard error (SE), minimum (Min), and maximum (Max).

Environmental Variables

The pore–water characteristics showed variability concerning mangrove ecotype, climate season, and lagoon zones. The highest salinity was recorded in the pore water from basin mangroves during the dry season (51.7 ± 17.7 psu), while the lowest salinity was measured in peten mangroves during the nortes season (21.2 ± 5.6 psu). However, there were no differences in pore–water salinity among mangrove ecotypes or seasons (**Supplementary Table A**). Regarding the location of the mangroves in the lagoon spatial gradient, salinity was significantly different in the inner, middle, and outer zones ($F_{2,34} = 28.45$, $p < 0.0001$) (**Supplementary Table B**).

The temperature does not show differences among mangrove ecotypes; however, with values ranging from 26.7 ± 0.7°C in nortes to 30.4 ± 0.7°C in the dry season, the main effect is due to climate seasons ($F_{2,40} = 99.7$, $p < 0.0001$). Spatially, the temperature does not show differences.

The pH values vary from 6.87 in fringe mangroves to 6.55 in peten mangroves. Therefore, pH decreases from the shore of the lagoon to inland. Considering the zones of the lagoon, the pH values ranged from 6.63 in the inner zone to 6.92 in the outer zone; with these values, the three zones were significantly different ($p = 0.0091$).

The redox potential remains with similar values among ecological types, varying from −184.39 mV in basin mangroves to −219.59 mV in peten mangroves. The seasonal gradient shows

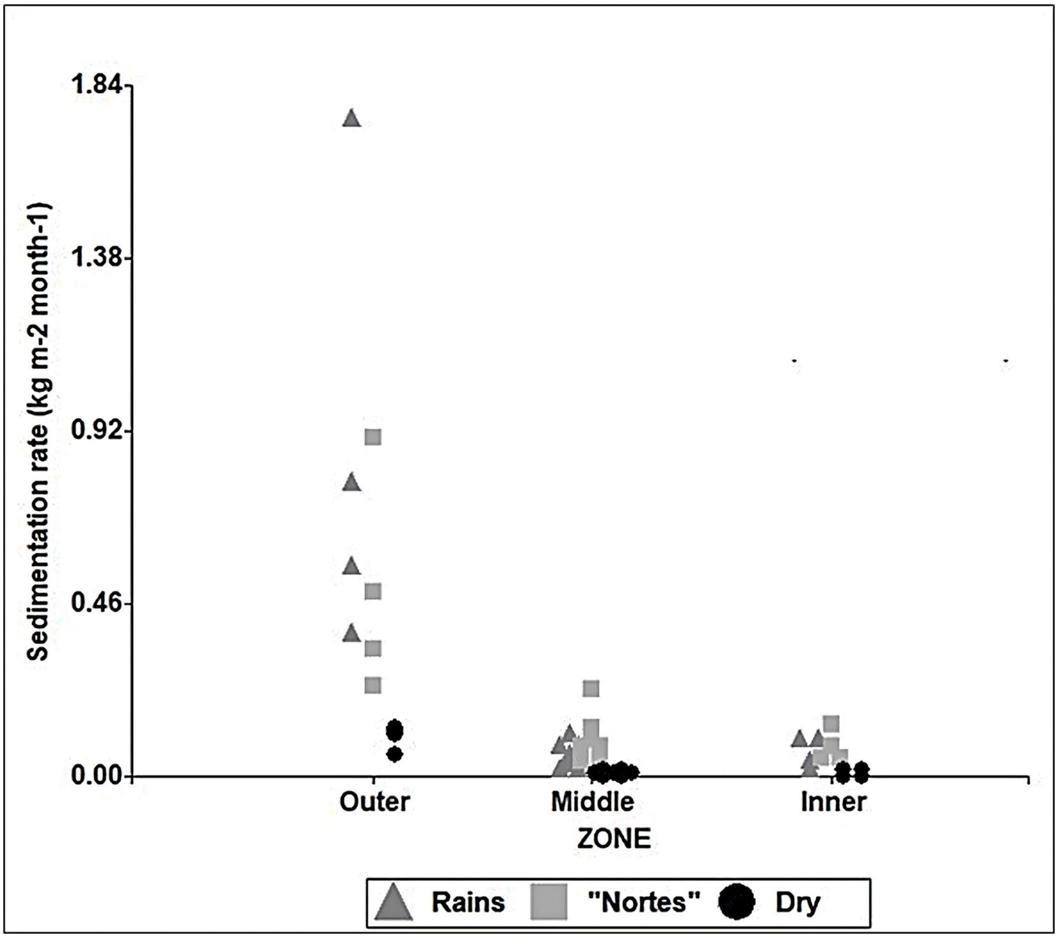

FIGURE 2 | Sedimentation rate (kg m^{-2} $month^{-1}$) during three seasons (rains, nortes, dry) according to lagoon zones (Outer, Middle, Inner) in Celestun mangroves.

reduced electron conditions in the sediments considering the redox potential and negative values (**Supplementary Table A**). Concerning the spatial distribution, the redox potential varies from −234.82 in the inner zone to −156.42 in the outer zone (**Supplementary Table B**).

The flood levels change according to the climate seasons. During the dry season, it was significantly lower than that during the rainfall and nortes seasons ($F_{2,19} = 3.56$, $p = 0.0486$) (**Supplementary Table A**). The significant differences are also observed between flood levels, which are higher in the inner than in the middle and outer zones of the lagoon ($F_{2,25} = 4.66$, $p < 0.0201$) (**Supplementary Table B**).

Sediment Characteristics and Source

According to the ecotype, the mangrove surface sediments show differences in BD, organic matter, total phosphorous, total nitrogen, and total carbon content. Although fringe mangrove sediments are denser (0.20 ± 0.16 g cm^{-3}), there are no differences among ecotypes. In most of the cases, more than half of the sediment content was organic matter (52.49 ± 16.55%); however, the highest proportion was found in peten mangroves (79.64 ± 3.53%), with significantly higher values than those of other mangrove ecotypes ($F_{3,86} = 10.27$, $p < 0.0001$). The total carbon and nitrogen values showed similar differences, significantly higher in peten mangroves than in the other ecotypes (**Table 2**).

The sediment characteristics vary according to their location around the lagoon. The sediments have a higher BD and lower organic matter (OM) content in the outer zone than in the middle and inner zones, while the highest total nitrogen and carbon contents were recorded in the inner zone (**Supplementary Table C**).

Regarding the isotopic composition of sediments, $\delta^{13}C$ varies from −28.37 to −22.61‰. The dwarf mangroves have significantly higher $\delta^{13}C$ (−22.65 ± 0.41‰) than fringe, basin, and peten mangroves. $\delta^{15}N$ is significantly higher in peten mangroves (7.85 ± 0.45‰) than in the other mangrove types. The N:C ratio did not present significant differences between mangrove types (**Table 2**).

The isotopic composition shows differences in $\delta^{13}C$, being higher in the sediment of the outer zone (−26.37 ± 0.09‰) than

TABLE 2 | Characteristics of the sediments in four ecological types of mangroves.

Mangrove type	BD ($g\ cm^{-3}$)	OM (%)	TP (%)	TN (%)	TC (%)	OC (%)	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)	N:C
Fringe	0.20	47.22[a]	0.11	1.21[a]	23.92[a]	19.77[a]	−27.01[a]	5.96[a]	0.06
	(0.02)	(2.61)	(0.01)	(0.11)	(1.05)	(1.17)	(0.29)	(0.32)	(0.01)
Basin	0.17	56.84[a]	0.07	1.34[a]	24.41[a]	21.77[a]	−27.07[a,b]	4.56[a,b]	0.07
	(0.02)	(2.48)	(0.01)	(0.11)	(1.05)	(1.17)	(0.17)	(0.18)	(<0.01)
Dwarf	0.14	45.85[a]	0.07	1.51[a,b]	24.41[a]	20.59[a]	−22.65[c]	3.76[b]	0.05
	(0.03)	(3.46)	(0.01)	(0.12)	(1.17)	(1.30)	(0.41)	(0.45)	(0.01)
Peten	0.12	76.64[b]	0.11	1.94[b]	34.44[b]	32.97[b]	−28.36[b]	7.85[c]	0.05
	(0.05)	(5.39)	(0.02)	(0.22)	(2.09)	(3.23)	(0.41)	(0.45)	(0.01)
p		**<0.0001**		**0.0248**	**0.0003**	**0.0001**	**<0.0001**	**<0.0001**	

BD, bulk density; OM, organic matter; TP, total phosphorus; TN, total nitrogen; TC, total carbon; OC, organic carbon, $\delta^{13}C$, $\delta^{15}N$, and N:C ratio. The values are means, and in parentheses, standard errors. The different characters indicate significant differences ($p < 0.05$) in ecological types.

in the inner and middle zones. The $\delta^{15}N$ and N:C ratios did not differ between zones (**Supplementary Table C**).

According to the $\delta^{13}C$ isotopic composition and the mass balance, the carbon source in mangrove sediments is associated with 92% mangrove leaves; most of the sediment components are produced *in situ*. In fringe, basin, and dwarf mangroves, these proportions are 96, 97, and 90%, respectively, while in peten mangroves, they are more than 99%. **Figure 3** shows the isotopic signature of different carbon sources and how mangrove sediment is distributed.

Sedimentation Controls

The SIMPER analyses indicate that dissimilarity (59–99%) between mangrove ecotypes is due to tree density (ind ha^{-1}). According to the Pearson correlation coefficients, the tree density correlates with mangrove structural vegetation characteristics (DBH, height, and basal area) and waterflood level. The physical–chemical variables and salinity have inverse relationships with DBH, height, basal area, and sediment carbon content concerning pore water. The sediment characteristics and OM percentage had an inverse relationship with BD and a direct relationship with total carbon and total nitrogen contents ($p < 0.0001$). Based on these relationships, the selected variables for LDA were tree density, pore-water salinity, flood level, and sediment OM content. According to these variables and sedimentation rates, the first axis in the discriminant space separates dwarf mangroves (CA1 $= -7.65$) associated with tree density. In contrast, the second axis shows a gradient of mangrove ecotypes associated with salinity, although with significant overlap between fringe, basin, and peten mangroves (CA2 $= -1.11$) (**Figure 4**).

DISCUSSION

Sedimentation Rates Related to Spatial and Other Patterns

The net sedimentation rate is higher in fringe mangroves than in basin, dwarf, and peten mangroves (**Figure 2**). This pattern is similar to the values reported by Adame et al. (2010) in Australian mangroves, where they found that geomorphological features influence sedimentation patterns. The geomorphological influence is reflected in the low values of sedimentation rates obtained in our study (0.001–0.59 g cm^{-2} $year^{-1}$), performed in the karstic environment of Yucatan Peninsula, and compared with those reported for mangroves growing in a geomorphologic scenario with rivers and tides up to 3 m, such as the Gulf of Papua (1.1–6.5 g cm^{-2} $año^{-1}$) (Walsh and Nittrouer, 2004).

The lack of statistical significance among the sedimentation rates of mangrove ecotypes is related to their high seasonal and spatial variabilities observed in different lagoon zones in Celestun. This pattern in sedimentation is associated with flood level, which is minimum in peten mangrove and increases to fringe mangrove (**Supplementary Table A**). In this karstic ecosystem of the Yucatan Peninsula, groundwater discharges and surface water flow should also be considered because the contribution of groundwater is of such magnitude that it changes the structure and function of mangroves, such as in mangroves of the peten type, structured around a point source of groundwater that emanates and flows toward the coast, transporting nutrients and particles from production and decomposition processes. In the mangroves of Celestun, these particles settle according to the characteristics of slope, energy, tides, and waves; in other words, local hydrology affects the transport and production of particles and how they become part of the sediment (Walsh and Nittrouer, 2004).

Concerning the spatial gradient, mangroves in the inner zone of the lagoon are influenced by freshwater discharges, which is showing a high amount of sediment composed of small-sized particles giving a relatively low BD compared with mangroves in the middle and outer zones (**Supplementary Table B**). Remarkably, the outer zone has a higher sedimentation rate due to having direct contact with the sea, receiving sandy sediments that usually have higher BD. This zone is directly influenced by the tidal dynamics of the Gulf of Mexico, which has an average tidal range of 35 cm (UNAM—Servicio Mareográfico Nacional, 2016) but may range up to 76 cm on the coast of

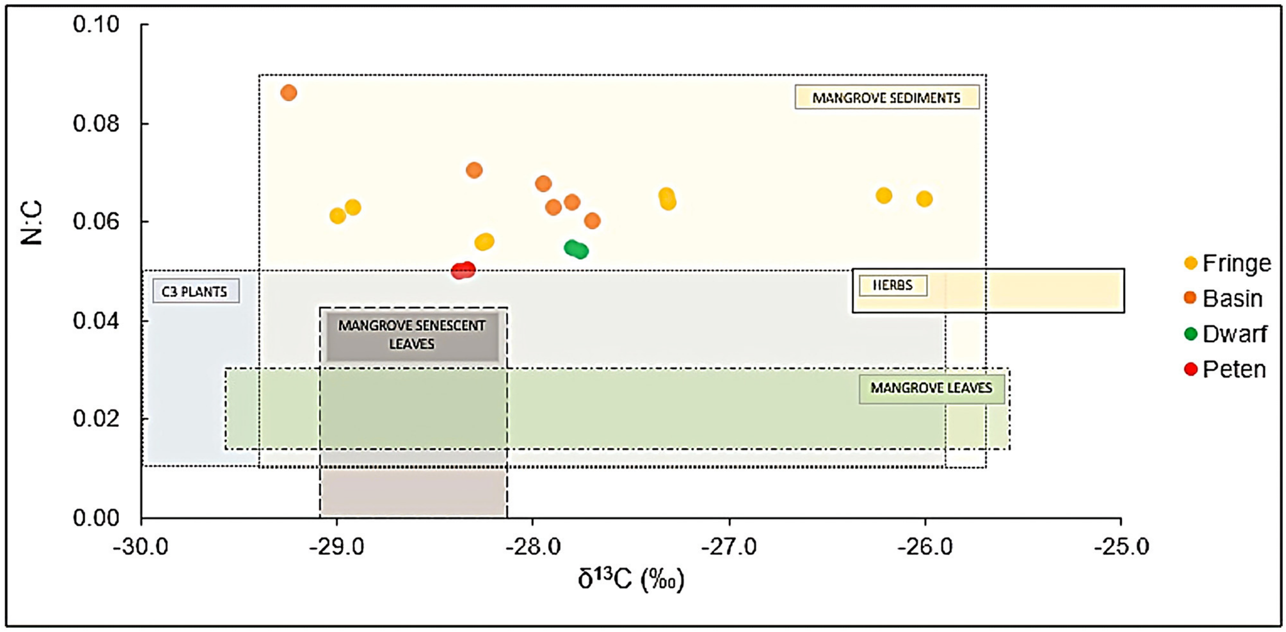

FIGURE 3 | Isotopic signals ($\delta^{13}C$ and N:C) in sediments of mangrove ecological types in Celestun. The values of possible sources of carbon and nitrogen are included: Reference data are from mangrove leaves (Wooller et al., 2003; Gonneea et al., 2004; Fry and Cormier, 2011), senescent mangrove leaves (Gonneea et al., 2004), mangrove sediments (Monacci et al., 2011), C3 plants (Adame and Fry, 2016), and mangrove associated herbs (*Batis maritima* and *Salicornia virginica*) (Grijalva, 2015).

Celestun Lagoon (Torres-Mota et al., 2014). These characteristics are responsible for the differences in sedimentation rate among lagoon zones. In the outer zone, sandy sediment inputs are strongly related to the relatively intense hydrodynamics in this zone, which is close to the sea. In the middle and inner zones, the hydrodynamics is much lower (Herrera-Silveira and Comin, 1995) because the influence of sea tides does not reach these zones but has a significant freshwater influence from spring inputs. This influence is also reflected in the higher OM content observed in the inner zone (**Supplementary Table B**).

On the other hand, the measurements performed in different seasons indicate that temporal variability is associated with hydrologic dynamics, with the surface flood water level as an indicator (**Supplementary Table A**). During the rainy season, the sedimentation rate is generally higher (**Table 1**) due to relatively higher water flows carrying sediments from inland to coastal zones due to the high freshwater supply and flood level (Zaldívar-Jiménez et al., 2010). However, there is a continuum of events from the rainy season to the nortes seasons. First, the fringe and basin mangroves increase their sedimentation rate, whereas dwarf and peten mangroves do so during the nortes season. This behavior could be related to these mangrove ecotypes' connectivity in the landscape and how the water flows from inland to mangrove fringes. This hydrological connectivity is essential because the water comes from the land within and from the effluents of the peten mangroves, and on a slope, moves through the shallow and basin mangroves until it reaches the fringe mangroves. In this displacement, the salinity and the availability of nutrients that sustain each ecological type and guarantee more environmental services per unit area are modified. For this reason, it is crucial to consider that the hydrological dynamics respond to the frequency and intensity of regional rainfall due to their connection with groundwater discharges of the springs and runoff. During rainy and nortes seasons, the water table rises, and springs in the area show the highest water discharges (Stalker et al., 2014), increasing the Wrunoff and sediments distributed in all ecological types of mangroves.

These water dynamics in the mangrove ecotypes throughout the lagoon areas and variability across the seasonal scale are reflected in the physicochemical characteristics of the pore water, such as salinity, pH, and redox potential (**Supplementary Tables A,B**), and with nutrient inputs. For example, relatively more acidic conditions occur in inland mangroves, such as dwarfs and petenes, while in the inner zone of the lagoon, the redox potential is more negative. More negative redox potential values mean that the flooding time is probably extensive, and OM degradation occurs in the absence of oxygen. These processes also affect organic carbon stocks registered in different mangrove ecotypes (Cinco-Castro, 2022). On the other hand, acidity is lower in fringe mangroves and the outer zone of the lagoon due to the influence of seawater on this variable. If we add the temporal variability, the results are that during the nortes season, the fringe and basin mangroves show higher pH, which should be related to local hydrology, as seawater inputs and turbulence favored by

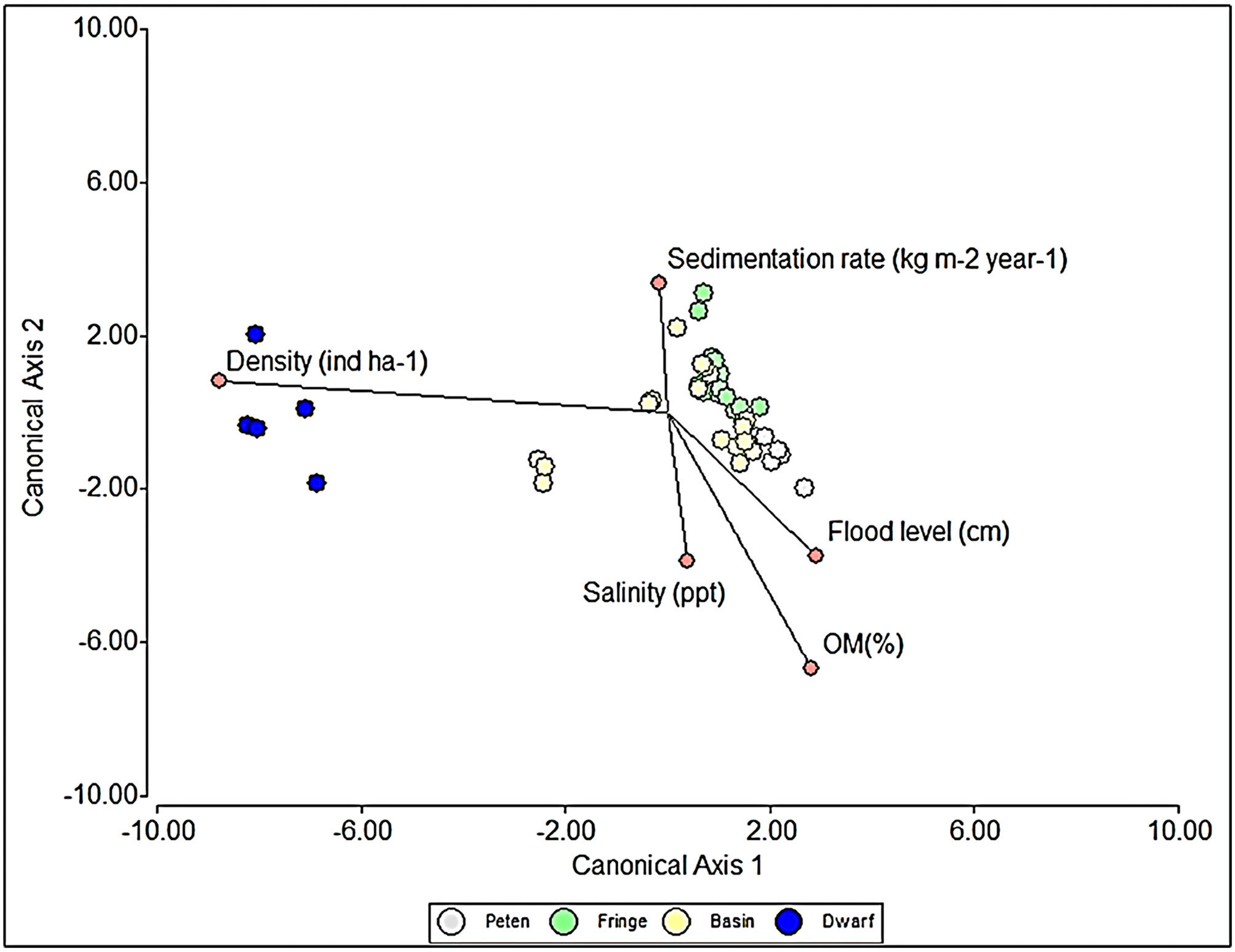

FIGURE 4 | An LDA. The distribution of sampling sites in four ecological types of mangroves (fringe, basin, dwarf, and peten), according to the density of trees, salinity, flood level, organic matter content, and sedimentation rate.

meteorological phenomena frequency along the coast during this period.

There is a relationship between forest vegetation structure and physicochemical variables of pore water. Salinity shows the same pattern, where it is relatively high in the outer zone due to marine influence and low in the inner zone due to water inputs from springs. Therefore, mangroves in the inner zone and peten show a better structure according to DBH and height (see Results section) than other ecotypes and other zones. This structure could be related to the low salinity in these zones (**Supplementary Table A**), where freshwater springs with high nitrates are dissolved, reducing vegetation stress and favoring biomass growth (Cintron, 1982; Herrera-Silveira, 1994). Then, the highest tree and seedling densities in the middle zone are due to the mixing characteristics reflecting an environment observed in estuarine conditions, which favors a significant interaction among species compared with the inner and outer zones. This interaction among mangrove species (*Rhizophora mangle*, *Avicennia germinans*, and *Laguncularia racemosa)* favors the observation of higher biomass production (Camacho-Rico, 2018; Bai et al., 2021).

All of the above factors are especially important due to directional relationships in different directions of the ecosystem, as observed among forest vegetation structure, hydrodynamics (flooding time, water sources), and sedimentation rates. These characteristics allow the evaluation of the condition and functioning of the ecosystem and how it could respond to sea-level rise. The areas with more significant sediment accumulation and exposed to constant changes in the tide, such as the outer zone, could be less vulnerable to these impacts from climate change if the sediment accumulation velocity is greater than sea-level rise. Knowing the relationship among ecological processes related to a specific ecosystem service is a pending issue in mangroves, and it must be approached with greater scientific rigor.

Much has been written about the interaction between structure, function, and ecosystem services (Fu et al., 2013); for example, the dynamics of water in the ecosystem is a process that translates into the provision of ecosystem services such as the provision of clean water and irrigation or that indirectly influences food production, the regulation of the microclimate or that favors aesthetics of the site. Something similar occurs with the carbon cycle as an ecological process from which services such as gas and temperature regulation, soil formation, and biodiversity conservation are derived. However, in mangroves, few examples (Lee et al., 2014) show this relationship between ecological processes and ecosystem services using direct measurements, remarking on the importance of this document.

Additionally, it is necessary to remember that sedimentation rates are related to hydrodynamics and that water sources and flows influence local hydrodynamics. However, precipitation, temperature, and cyclone frequency could also be considered sources of variation in hydrologic dynamics, and they explain the forest vegetation structure (Simard et al., 2019) and their variability at different spatial scales. Furthermore, climate change could affect mangrove ecosystems when current precipitation and sea-level rise patterns are modified (IPCC, 2021), modifying the hydrologic and sedimentary dynamics as sediment exportation and accumulation. If the climate and sea-level change, different scenarios of mangroves identified in this study could be changed, endangering their conservation, and consequently, the ecosystem services related to each one.

It is crucial to consider that the sediment accumulation rate measured in this study represents the net rate, and it is necessary to consider how water fluxes change spatially and temporally. The results integrate the exposition of the traps to the field conditions, sediment resuspension, and transport in each zone according to their specific hydrodynamic. This study represents a baseline for tracing sediment accumulation; however, it is necessary to implement monitoring programs that permit us to register the variability of sedimentation rates and their relationship with variables in a long-term context to determine changes that could be alarming and to implement management policies.

Sediment Composition and Source

The sediment composition is related to the local hydrodynamics and the sources of the materials that form the sediments. In the inner zone of the Lagoon, the sediment is mostly lime, while in the outer zone, it is sandy. The inverse relationship between BD and organic matter (OM) content expresses the spatial heterogeneity of sediment composition in Celestun (**Supplementary Tables B,C**). The BD of the sediment in Celestun mangroves (0.12–0.20 g cm^{-3}) is lower than that reported for other mangroves (0.7–1.42 g cm^{-3}), but our OM values (45–76%) are higher than those reported from other sites (from 0.65% to 22.89%) (Hossain and Nuruddin, 2016). From this, two issues should be highlighted: (1) the inverse relationship between BD and OM content is evident, and (2) the differences between our data and global data reported in the literature are likely related to the variability of local characteristics. Similar differences are observed contrasting our data of mangroves in the Yucatan Peninsula with the sediment of other Mexican mangroves (BD: 0.9–0.22 g cm^{-3}; OM: from 6.9% in deep layers to 85.8% in superficial layers) (Moreno et al., 2002).

The increasing BD and decreasing OM in the sediment of the mangroves from the inner to the outer zones of the lagoon are also related to the mangrove vegetation structure. The highest OM contribution occurs in the inner zone, where the freshwater inflows are high, and tree height shows the maximum values and structures near the soil level as proper roots and pneumatophores. These structures directly influence sediment accumulation, diminishing the water velocities. Additionally, OM is high in the middle zone due to estuarine water conditions, which favor overlapping different mangrove species (*R. mangle*, *A. germinans*, and *L. racemosa)* and favor litterfall production and OM input to the sediment (Camacho-Rico and Herrera-Silveira, 2015). In summary, the differences in sedimentation rates among the lagoon zones are explained by a smaller size of the sediment particles, and consequently, lower BD and higher OM in the inner zone than in the outer zone, where marine influence and sandy materials dominate.

The $\delta^{13}C$ differences among mangrove ecotypes are related to their hydrology. The significantly higher mean value in dwarf mangroves can be related to the C4 plant association, such as *Spartina sp.* (from −10‰ to 0‰), and due to the accumulation of carbonates in freshwater (from −20‰ to 14‰), which is usually present in this mangrove ecotype (**Table 2**). Regarding spatial distribution, the inner zone shows low values (−27.52 ± 0.08‰) due to the strong influence of continental organic material as terrestrial plants. The outer zone has the highest values given by marine influence, probably by phytoplankton and seagrasses (**Supplementary Table C**).

Then, the isotope results show that carbon sources in mangroves, without river influence, are mainly autochthonous, and management strategies could be locally applied. Like the ones in this study that do not receive surface water flows from inland areas, mangroves have productivity mainly stored in the same place where they are produced, so they are functionally a sink of sediments, nutrients, and particulate organic carbon. Therefore, the management strategies must be focused on avoiding wastewater inputs to the mangrove forest because this would modify the nutrient dynamics by changing the ecosystem's structure due to variations in salinity and hydroperiod, for example. Additionally, logging and land-use change should be avoided because the carbon that accumulates and forms part of the sediments would no longer have its origin in the coastal ecosystem. Local management of autochthonous material is essential due to its relationship with blue carbon stocks (Saintilan et al., 2013).

According to isotope results, $\delta^{15}N$ is a good indicator of pollution. High $\delta^{15}N$ (7.85 ± 0.45‰) in peten mangroves indicates anthropic nitrogen inputs (Bergamino et al., 2017) through sinkholes or springs associated with this mangrove ecotype. According to the spatial distribution, the inner and middle zones showed similar $\delta^{15}N$ values, indicating that these sites were related to nitrogen inputs with anthropic origins, so the water that reaches the mangroves has had contact with or comes from human activities. This information highlights the influence of local management of ecosystems and the importance of management with a basin approach for conserving mangroves, which is challenging to implement in a karst environment where the sources and flows of water are underground, and they present connectivity at different spatial scales.

In another order of ideas, accretion in Celestun fringe mangroves is 2.91 mm $year^{-1}$ (Cinco-Castro and Herrera-Silveira, 2020), while the sea-level rise in the Gulf of Mexico is 3.13 mm $year^{-1}$ [National Oceanic and Atmospheric

Administration (NOAA), 2016]. In this sense, the coastal area of the Yucatan Peninsula has changed since the Pleistocene. Among the main changes is the formation of ripples on the coastline that currently represent coastal wetlands with deposits of carbonate sediments from the quaternary (Bautista et al., 2005) that have continued to accumulate to the present day, conferring to coastal ecosystems their capacity to cope with sea-level rise (Kumara et al., 2010).

Conservation and Management Implications

The results in this study support the hypothesis that net sedimentation rates in mangroves are higher in zones with high hydrodynamism than in quiet zones. Therefore, restoring hydrodynamics is a significant action to stimulate sediment accretion and recovery of mangrove ecotypes that are better adapted to the influence of high hydrodynamism as fringe mangroves (Teutli-Hernández et al., 2020). However, organic carbon accumulation is also related to the origin and accumulation of OM in the sediments. The OM is higher where hydrodynamics is lowest and markedly contributes to soil formation, such as in the inner zone. In addition, other ecological functions and ecosystem services can be provided differentially by different mangrove ecotypes in a territory (Himes-Cornell et al., 2018). Therefore, mangrove types must be considered in mangrove management as part of the mosaic with a landscape approach since this heterogeneity adds richness and stability to the ecosystem.

At the interecosystem scale (water body–mangrove), the seasonal differences between seawater and freshwater flow determine the role of different mangrove ecotypes. This information can be helpful to establish mitigation and adaptation strategies in front of climate change and particularly to understand how mangroves from karstic regions could adjust to sea-level rise. The dwarf and peten mangroves may be essential in mitigating climate change due to their high organic carbon accumulation. The fringe and basin mangroves have a more efficient role in buffering sea-level rise and sea storm effects because of their higher sedimentation rates. According to the above, the management strategy should be oriented toward conserving the mosaic of mangrove ecotypes that form a heterogeneous landscape that functions as a continuum under hydrogeomorphological drivers (Twilley and Rivera-Monroy, 2009). Actions that favor sediment supply, water flows, and healthy forest vegetation structure should focus on improving, recovering, and even maintaining vertical soil accretion as a proxy of soil formation as a key ecological process for providing ecosystem services.

CONCLUSIONS

Mangroves in karstic zones have a lower sedimentation rate than mangroves submitted to other conditions, reflecting the importance of hydrodynamism on sedimentation patterns. Spatial and seasonal hydrological dynamics define the changes in the sedimentation rate of mangroves around Celestun Lagoon. Mangroves close to the coastal zone submitted to high hydrodynamism accumulate sand material with low organic content at high rates, thus protecting against sea storms. Mangroves in the inner parts of the coastal zone show lower sediment accumulation rates but with high organic content, thus contributing to buffering climate change in the long term. The spatial differences observed in the sedimentation rates and the provision of related ecosystem services should be considered to manage a sustainable and desirable mosaic of ecological types of mangroves in karstic zones.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

AUTHOR CONTRIBUTIONS

SC-C carried out field and laboratory work, analyzed data, prepared figures and/or tables, and authored and reviewed drafts of the article. JH-S and FC authored or reviewed drafts of the article and approved the final draft. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/ffgc.2022.733820/full#supplementary-material

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Stocks and Productivity of Dead Wood in Mangrove Forests: A Systematic Literature Review

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The functional and ecological importance of dead wood in terrestrial forests is widely recognized and researched. In contrast, much less is known about dead wood in mangrove forests, despite its known or demonstrated contribution to key ecological processes including nutrient cycling and seedling recruitment. In addition, mangrove dead wood provides an important service for millions of people; harvesting wood for fuel is widespread in mangroves and is often vital for the lives and wellbeing of people living close to these forests. Limited information on stocks and production, and the drivers of these, means that understanding and managing the supply of this service is difficult. Here we conduct a systematic review of the literature on dead wood stocks and production in mangrove ecosystems. Four hundred and seventy-five subject articles were found, with large gaps in geography, species, and forest type. After excluding records that were not relevant to our study and those from mass mortality events, 68 studies remained. We also added new data from 9 sites in Kenya, to provide overall estimates of mean ($\pm$ SD) stocks of dead wood of 16.85 ± 25.35 Mg ha^{-1} standing and 29.92 ± 36.72 Mg ha^{-1} downed. Our analysis shows that potentially, higher stocks of dead wood might be found in forests without evidence of human impact. Average mean production with 95% CI was 6.30, 3.10–11.40 Mg ha^{-1} yr^{-1}. Estimates of daily wood use were applied to give likely demands on wood from mangrove dependent communities. This review reveals the paucity of research on mangrove dead wood, hence these estimates of average stocks and productivity remain very limited and thus, further work on the dynamics of dead wood in mangroves and the ecological effects of its removal is needed.

Keywords: woody biomass, forest, mangrove, standing dead wood, downed wood, carbon, biomass

INTRODUCTION

Mangrove forests are communities of trees and shrubs found in the intertidal zone in the tropics and subtropics (Lugo and Snedaker, 1974). With a global area of 13.76 million hectares (Bunting et al., 2018), mangroves contribute about 0.3% of the world's forest cover (FAO, 2016). Mangroves provide numerous provisioning, supporting, regulating and cultural services to coastal populations and have been conservatively valued between USD33,000 and 57,000 per hectare per year (Sathirathai and Barbier, 2001). Despite their relevance, between 35 and 50% of the pre-industrial area of mangrove forest has been lost. Whilst global decline of these forests continues, the rate of loss has slowed to $\sim$0.16% yr^{-1} (Friess et al., 2019; FAO, 2020) possibly as a result of increasing attention to conservation and growing recognition of the role of mangroves as natural carbon sinks.

The Food and Agriculture Organization (FAO) defines dead wood (DW) as "All non-living woody biomass not contained in the litter, either standing, lying on the ground, or in the soil" (FRA, 2015). Dead wood has received increasing attention in terrestrial forestry over the past two decades, reflecting the growing knowledge of its importance in forest ecology. Many organisms rely wholly or partly on the presence of DW (Heilmann-Clausen and Christensen, 2004; Seibold and Thorn, 2018). For example, Siitonen (2001) found that DW habitats supported 20–25% of all forest dwelling species in Finland. In addition to directly providing habitat, DW influences nutrient cycling and retention, pedogenesis and plant recruitment dynamics in forests. Monitoring systems designed to measure and promote forest biodiversity, such as those adopted by the European Environment Agency, now use DW as an indicator of ecological quality (Söderberg et al., 2014) and research is devoted to ways of increasing DW quantity and diversity in forests (e.g., Christensen et al., 2005).

DW in mangrove forests has received much less research and policy attention, despite evidence of its importance in a range of ecological processes, including those found in terrestrial forests but also some of particular or sole relevance to tidal forests. For example Romero et al. (2005) studied DW decomposition and its contribution to nutrient dynamics at a Florida site which, like many mangroves, is subject to major disturbance from tropical cyclones that can result in sudden depositions of large volumes of DW. They found that such incidences have "profound" impacts on nutrient dynamics. In particular, downed wood can be a major source of nitrogen and phosphorus in forests that are limited by these nutrients.

DW is also an important component of the carbon stocks and flows in mangrove forests. Robertson and Daniel (1989) produced one of the few estimates of both stock and production dynamics of mangrove DW. In this study, carried out in Australia, the authors reported thata mature, mixed *Rhizophora* spp. forest had a mean aboveground DW stock of 14.2 Mg ha^{-1} (9.4 Mg fallen and 4.8 Mg standing), compared to 403 Mg ha^{-1} dry weight living biomass. Hence DW may be a significant part of the carbon stock in many mangrove forests; note that these figures do not include dead (belowground) roots. Buried DW may be a larger carbon stock than aboveground DW in many forests; for example Tamooh et al. (2008) found 32.6 Mg ha^{-1} of dead roots (compared with 35.8 Mg ha^{-1} live roots) in a mature Kenyan *Rhizophora* spp forest.

These ecological functions—mediating supplies of key nutrients such as nitrogen and phosphorus, supporting a range of fauna and acting as a store of carbon—are shared with terrestrial forests. In contrast, other processes which involve mangrove DW are unique or of special importance to these aquatic habitats. For example, DW can influence stream and tidal flow patterns, changing how aquatic fauna access and use the forest and influencing the accumulation of sediment (Allen et al., 2000). Mangrove propagules are dispersed by floating in the water; the passive trapping of propagules by woody debris is important in the recruitment of new trees and in the recovery of damaged or cleared areas (McKee et al., 2007).

In addition to its ecological importance, DW is a crucial resource for many human communities living in or adjacent to mangrove forests. Biomass remains the main source of fuel for billions of people; those close to mangroves often preferentially collect mangrove wood, for convenience but also because it has high density, burns at high temperatures and can produce desirable flavors (Huxham et al., 2017). For example, at Gazi Bay in Kenya, more than 70% of households rely on wood collected from local forests, including the mangroves, and use an average of 1.2 kg per capita per day, spending an average of 22 h per month collecting this wood (Jung and Huxham, 2018). Understanding the importance of this ecosystem service requires sensitivity to the social and cultural context. The labor of fuelwood collection in Africa is performed overwhelmingly by poor women and girls. Standard economic assessments may underestimate the value of this fuelwood provision and the opportunity costs, such as time not spent studying or at school, suffered by the girls (Huxham et al., 2015). Cash income is strongly correlated with mangrove fuelwood use. In a review of case studies of mangrove communities from around the world, Huxham et al. (2017) found $\sim$ 90% of households using mangrove wood in Vietnam, Indonesia, The Gambia and Cameroon, whilst none reported using mangrove wood for fuel in a Mexican study, where people could afford alternatives. Hence, supplies of fuelwood (which is mostly but not exclusively dead) are vital resource for some communities and irrelevant for others. So, understanding and managing supplies of mangrove DW to dependent communities requires understanding of cultural, social and gender contexts. Wood collection is a significant driver of mangrove degradation and destruction. Chowdhury et al. (2017) found that wood collection, including for fuel use, was implicated in 44% of the cases of degradation that they studied globally and in 90% of those cases from Africa. Therefore, the supply of DW from mangroves has important implications for the health and welfare of millions of people and for the conservation of mangrove forests.

At present, there is limited information on the stocks, and less on the productivity, of DW in mangroves and on the factors that drive these variables (Sitoe et al., 2014; Kauffman et al., 2016). To our knowledge there are no published attempts at estimating what might be sustainable levels of DW harvest from a mangrove forest, despite the importance of this for mangrove conservation in most countries that support mangroves. In this study we aim at filling this research gap by pursuing the following objectives: (a) conducting a systematic literature review to identify the current estimates for above-ground DW stocks and productivity in mangroves and explore the influence of potential drivers, including location, species and impacts of human use, on DW; (b) comparing literature values on stock, and different methods for estimating productivity; (c) exploring the possible implications of DW estimates for management of mangrove forests.

METHODS

We used four methods to collect data on above-ground mangrove DW stocks and production: (a) a systematic review of published literature; (b) incorporation of any relevant data and papers used in the IPCC (2014), which provides default values for carbon assessments including of DW under IPCC Tier 1 assumptions, which were not already included during the initial review; (c) analysis of published forest data on mangroves in Kenya, collected by members or colleagues of the current team using consistent methodology at all sites; (d) search and analysis of published mangrove forest data on publicly accessible databases. Key aims were to estimate the average DW stocks and production in mangroves, along with the variance around these averages, to understand the influence of likely ecological and anthropogenic drivers of these variables and to identify research gaps and/or inconsistencies.

Systematic Literature Review

Search Strategy and Identification of Relevant Studies

Four search engines including the Institute for Scientific Information (ISI) "Web of Science," CAB Abstracts, ProQuest and JSTOR were used to find potentially relevant publications, which were either written in English or included an English abstract. There was no limitation on date of publication. The initial search term was:

> "*TOPIC: (mangrove* AND (biomass OR carbon OR productivity OR production)) OR TOPIC: ((dead wood OR standing dead wood OR downed wood) AND (biomass OR carbon OR production OR productivity)) AND TOPIC: (mortality) AND TITLE: (mangrove* AND ((dead wood OR standing dead wood OR downed wood) OR (biomass OR carbon OR production OR productivity)))*"

A total of 9,754 articles were found with this inquiry. A quick inspection showed that most of these articles were irrelevant given the versatility in use of the terms "*production*," "*productivity*" and "*mortality*"; therefore, the search was refined by excluding most research areas, leaving the following core disciplines: "*Environmental Sciences and Ecology, Water Resources, Fisheries, Marine Freshwater Biology, Meteorology, Atmospheric Sciences, Geography, Biodiversity Conservation, Social Sciences and Other Topics, Forestry, Energy Fuels, Plant Sciences, Oceanography, and Social Issues*."

Screening and Eligibility Criteria

The refined search resulted in 8,252 articles (**Figure 1**). Articles whose titles were relevant where selected and their abstracts were all independently read and categorized. Duplicates were excluded and the remaining articles were categorized into three groups: "definitely relevant," "possibly relevant" and "definitely not relevant." The "definitely relevant" class contained articles where the abstract clearly mentioned DW stock and/or production. The "possibly relevant" group mainly consisted of papers with mention of mangrove biomass and/or carbon, litter production and standing litter. As mangrove biomass and carbon studies, and litter production studies, usually focus on live biomass and leaf litter rather than wood litter, respectively, it was generally not possible to determine the relevance of these articles by only reading the abstract. Categorizations made by individual readers were compared to check for consistency; there was conformity in all the categorizations between readers.

A total of 283 articles were categorized as "definitely relevant" and "possibly relevant." The reference lists of these were then scrutinized for any additional relevant studies that had not been identified during the initial literature search. A further 192 articles were found to be possibly relevant and were available online. Thus, 475 articles were read in full for relevant data and information.

Data Extraction and Quality Assessment

During the full-text reading, data on the following quantitative and qualitative variables were collected: location (country, site, and coordinates), dominant mangrove species, stock of total DW (standing and/or downed), basal area and density of standing DW, biomass of standing dead and downed wood, production of DW, mangrove tree mortality rate and wood decomposition rate. Additionally, any evidence of human impact (such as wood removal activities including fuelwood collection or shrimp production) and management status, the type of forest (planted or natural stand) and the methodologies used in the studies were recorded. Data from studies undertaken after mass mortality events such as typhoons were excluded since these events can result in sudden, very large and unrepresentative stocks of DW (Stephenson et al., 2011). Similarly, articles that only covered litter production were set aside. Eventually, 67 of the 475 articles were found to have data relevant to our Research Topic.

Data Summary and Analysis

Retrieved data were converted to common units of measurement: Mg dry mass ha^{-1} and Mg dry mass ha^{-1} year^{-1} in the case of DW stock and production data, respectively. Where information was given on downed wood volume these data were converted into mass (Mg ha^{-1}) using wood density values provided by Simpson (1996). Where basal area (m^2 ha^{-1}) and stem density (stems ha^{-1}) were given the allometric equation provided by Clough and Scott (1989) was adopted to calculate standing DW (Mg ha^{-1}) using Equation 1.

$$\ln \ Biomass = A + B \times \ln D \tag{1}$$

where A and *B* were regression constants and varied between species and tree components, and *D* was the average DBH (cm) in the stand (or in case of basal area, it was DBH ha^{-1}) (Clough and Scott, 1989). **Table 1** gives some examples of A and B values used.

Tree biomass (Mg) was calculated and multiplied by the relevant stem density to get a unit area value (Mg ha^{-1}). DBH (cm) was calculated from basal area (Equation 2) where this was given; where it was not stated, but the average DBH for trees in a study was, this was applied to standing tree stock data.

$$Basal\ area = \pi (DBH/2)^2 \tag{2}$$

Production of mangrove DW was estimated using three approaches:

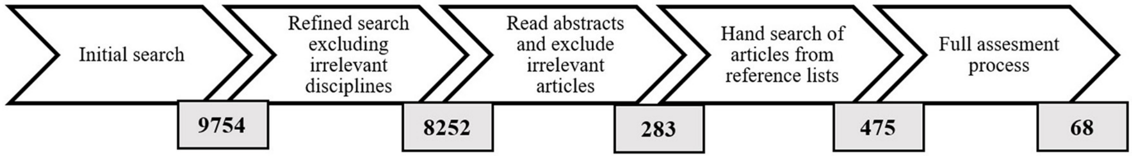

FIGURE 1 | Description of steps taken to find and refine literature along with the number of relevant publications found at each point.

TABLE 1 | Regression constants for calculating aboveground biomass of different mangrove species (Clough and Scott, 1989).

Species and tree component	A	B
Rhizophora apiculata, R. stylosa		
Branch	−1.8953	2.6844
Stem	−1.0528	2.5621
Bruguiera gymnorrhiza		
Branch	−1.5012	2.2789
Stem	−0.6482	2.1407
Ceriops tagal var. australis		
Branch	−1.7061	2.5516
Stem	−0.8333	2.3393

1) It was directly recorded where it was explicitly measured or calculated by the primary authors.
2) Where mortality rates of trees were given for a site these were used to infer DW production. Percentage mortality rates were used in combination with biomass values to calculate annual DW production rates (Mg ha^{-1} yr^{-1}). Where biomass was not provided, it was calculated from mean DBH (cm) using equation 1 (Clough and Scott, 1989) as indicated earlier for the case of stocks.
3) In some cases, papers did not directly give productivity or mortality rates but rather presented self-thinning or mortality models. Where appropriate relevant data on DBH/stem density etc was also presented, mortality rates were derived and used to calculate productivity as described in 2.

IPCC Wetlands Supplement

The most authoritative collection of information that is currently published on mangrove DW is the IPCC (2014) which lists studies that include default values of DW can be used in Tier 1 assessments of carbon stocks in mangroves. These papers were checked for any information in addition to that discovered during the literature review. This process added one additional paper to the list of those reviewed.

Additional Field Data

Three of the authors (ML, KJ, and HM) work to support the Mikoko Pamoja and Vanga Blue Forest projects which are two mangrove conservation initiatives based in southern Kenya (ACES, 2021). The projects collect monitoring data from 25 permanent forest plots, which include information on DW. These data were used to add three additional sites to this analysis. Further, data collected from six other mangrove sites in Kenya (Mugi and Kairo, 2021; **Figure 2**) were also used. These were collected using the same field protocols from temporary plots on various occasions between 2015 and 2020; the forest assessment methods described by Kauffman and Donato (2012) were applied. The studies covered 74% of the mangroves in Kenya where nine species have been documented in a horizontal species zonation typical of the Western Indian Ocean (WIO) region (Bosire et al., 2015; GoK, 2017). The sites are drawn from along the whole Kenyan coast. The most northerly sites are near Lamu (**Figure 2**), an area which contains > 60 % of the mangrove coverage in Kenya with relatively structurally complex forest formations (GoK, 2017). Further south, the low-lying estuarine system of Tana River (**Figure 2**) is dominated by distinctive stands of mangroves and associated species, and dwarf *Avicennia marina* stands on the landward (GoK, 2017). At the southern coast of Kenya, mangroves are dominated by mixed species stands (Mungai et al., 2019), with near pristine *Rhizophora* dominated stands on Sii Island at the southern-most part of the country (GoK, 2017). Human-induced losses and degradation of mangroves in Kenya have been widely reported (Kirui et al., 2013; Bosire et al., 2014; GoK, 2017; Mungai et al., 2019).

To calculate biomass at these sites, the bespoke equation developed by Cohen (2014) was used:

$$LN\ Biomass = -2.29711\ \left(\left(LNdbh\right)\ \times 2.54528\right) \quad (3)$$

Equation 3 estimates the biomass of standing live trees. It was also used in the estimation of dead tree biomass where it was combined with corresponding decomposition constants given by Kauffman and Donato (2012), where in standing dead trees in decay status 1 and 2, 2.5% and 15% were subtracted based on the leaf and branch biomass loss, respectively. The formula for conical volume was multiplied by species-specific wood density values obtained from Simpson (1996) to calculate the biomass of dead trees in decay status 3 (Equation 4).

$$B_{Status\ 3} = \left[\frac{\left(H \times \pi \times \left(\frac{DBH}{2}\right)^2\right)}{3}\right] \times q \quad (4)$$

where H is the tree height (in cm),

DBH is the diameter at 130 cm from the ground,

and, *q* is the wood density: 0.867, 0.780, 0.803, 0.741, 0.700 and 0.661 g cm^{-3} for *R. mucronata, S. alba, C. tagal, B. gymnorrhiza*, *Xylocarpus granatum* and *A. marina*, respectively (Simpson, 1996).

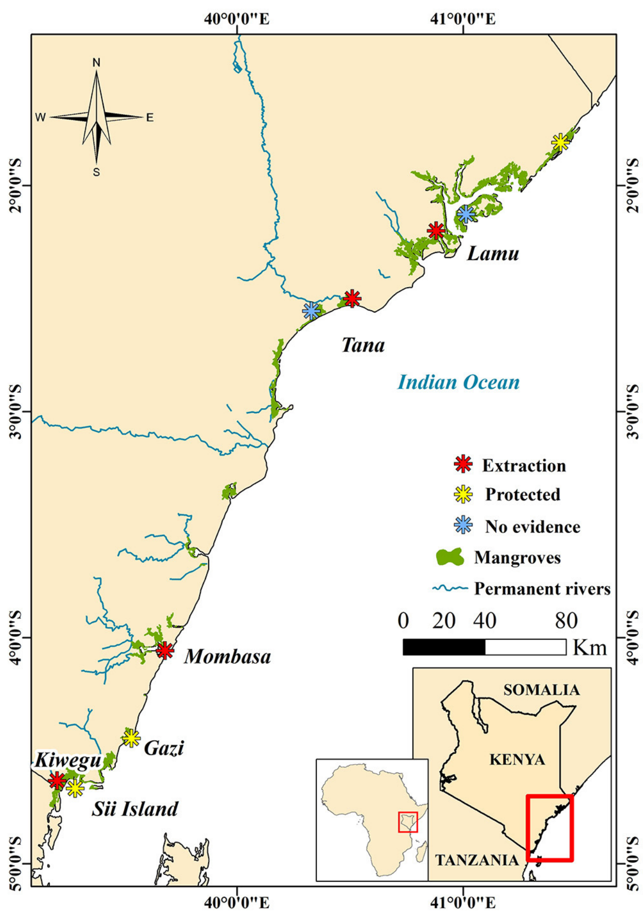

FIGURE 2 | Study sites for mangrove dead wood stock in Kenya.

Swamp and GlobAllomeTree Database

SWAMP is a database containing information from mangroves and peatlands across 27 countries; most of these datasets are publicly accessible and some include data on DW (SWAMP Database Management – SWAMP, n.d). GlobAllomeTree is a web platform for sharing and providing access to tree allometric equations, including mangroves. All relevant, accessible datasets and equations were scrutinized for additional data, but no sites not already included in the other searches were found.

Statistical Analyses

Statistical analyses were conducted using Microsoft Excel and R statistical software (version 4.0.2). Summary statistics were applied to characterize data retrieved from literature and those added from our local sites. Stocks of dead wood and productivity were compared between forests with and without evidence of human impact using ANOVA or non-parametric equivalents where appropriate.

When estimating average productivity, the paucity of data and uncertainty about underlying mechanisms and statistical distributions suggested caution when using parametric statistics. Hence bootstrapping (with the "boot" function in R) was used to produce an average (with non-parametric 95% CIs).

RESULTS

Study Characteristics

The refined literature search revealed that the first studies written on mangrove DW were published in the 1930s with a gradual increase in research on mangrove DW in the 21st century. Between 1932 and 2000, there were 254 scientific works published on mangrove dead wood, and since 2000, this number quadrupled. However, further investigation showed that most of these publications were irrelevant to the present study as they did not cover stocks or production rates of DW. The 68 articles found to be relevant to our study (67 from "the literature search" and an additional one from the IPCC wetlands supplement) were published between 1978 and 2021 (**Figure 3**, **Appendix 1**), with only 8 published in the 20th century. All subsequent discussion of published literature concerns only these 68 articles.

Relevant studies came from 38 countries, with Africa and Asia each having eleven (11), seven (7) in the Americas, three (3) in the Caribbean, five (5) in Oceania and one (1) in the Middle East (**Figure 4**, **Supplementary Table 1**). Twenty (20) of the publications reported on multiple sites while the rest were conducted at single sites, although larger forests were often stratified into different areas. As can be observed from **Figure 4**, the publications represented a relatively small area of the global mangrove coverage; from the 118 countries with mangroves (Giri et al., 2011), 32 % were represented.

A majority (95%) of the publications reported on primary studies undertaken in the respective sites in either temporary or permanent sample plots. The planar intersect technique (Van Wagner, 1968; Allen et al., 2000) was the most common procedure for assessing stocks of downed wood whereas forest productivity assessments reported wood litter from the litter traps technique. Cases of standing DW used tree measurement techniques described by Kauffman and Donato (2012). Only one of the publications (Steinke et al., 1995) reported use of a destructive sampling method to measure DW, which involved removal and sorting of above ground living and DW material from sampling plots. A summary of the data on DW production, standing dead and downed wood in mangrove forests is provided as **Supplementary Table 1**.

Mangrove DW stocks were reported as standing dead, downed, total DW, or a combination of these. Data on stocks of downed and standing wood were more frequent (34 and 25 articles, respectively) as compared to reports of total DW stock for which only 9 articles could be found from the search.

An indication of human impact and/or the management status of the forest stands was given in 44 articles (although in most cases information was scanty and qualitative, for example a comment in the site description that a forest "showed signs of cutting" or "was a protected area"). Such information was used to categorize stands as "protected" (where this was stated explicitly and there was no contradictory information, for

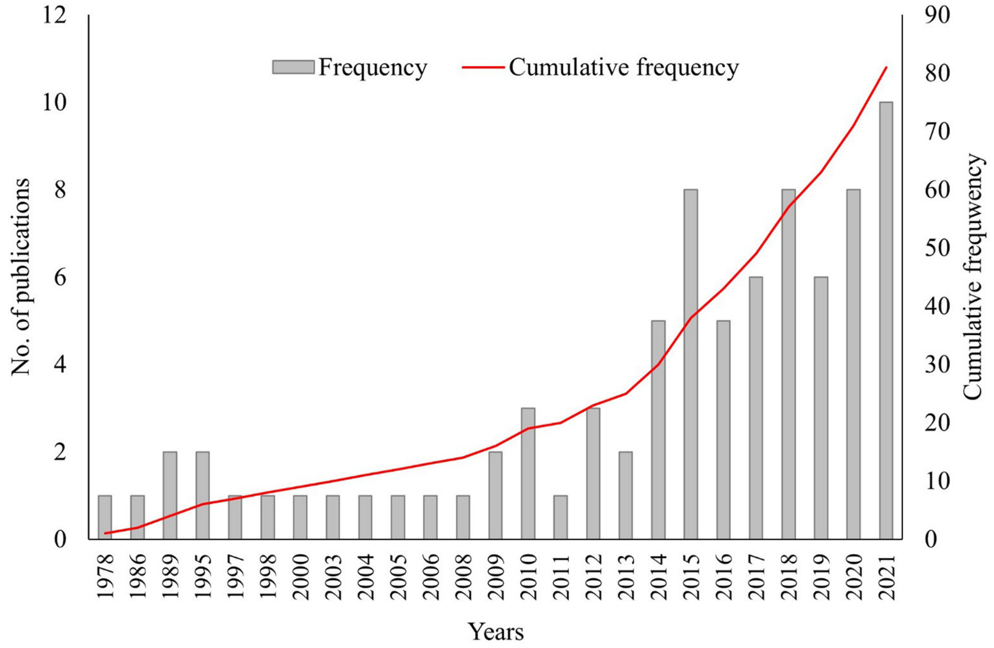

FIGURE 3 | Published research in mangrove dead wood over time.

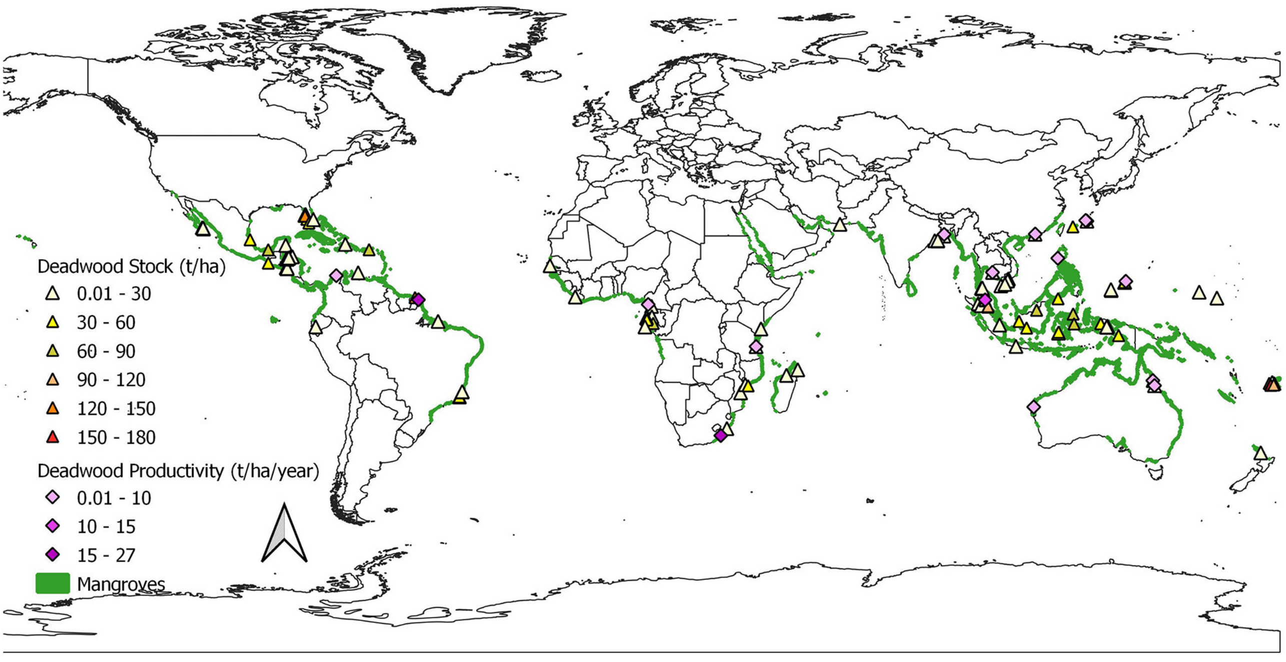

FIGURE 4 | Study sites for dead wood stocks and production. Stock values are a combination of total, standing, and downed wood biomass.

example stating protection was not enforced), "exploited" (where this was explicitly stated in the article) or "no evidence" (where no information on levels of human impact was given). Twenty-four articles described their sites as either protected areas with prohibited deforestation or as having no evidence of mangrove wood removal/human impact. Extraction of wood for fuel and/or timber, and to support fisheries or shrimp aquaculture were the most frequent reported land uses. The least described forms of use included exploitation for medicinal purposes, oil palm plantation and salt production.

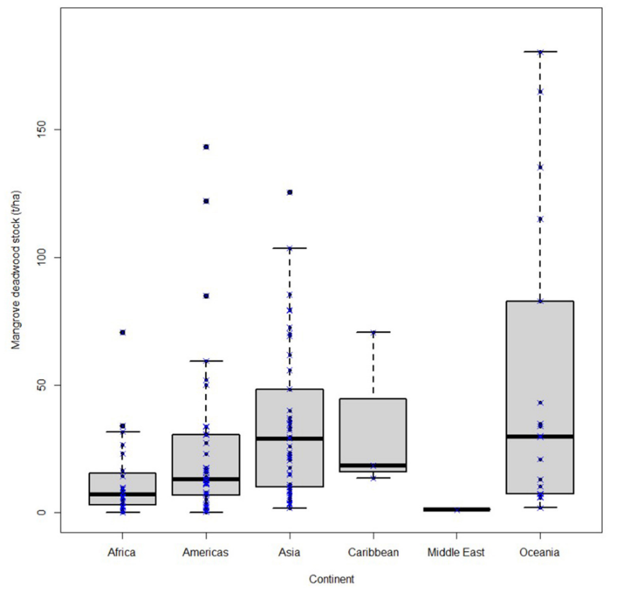

FIGURE 5 | Mangrove dead wood stocks Mg ha^{-1} across regions.

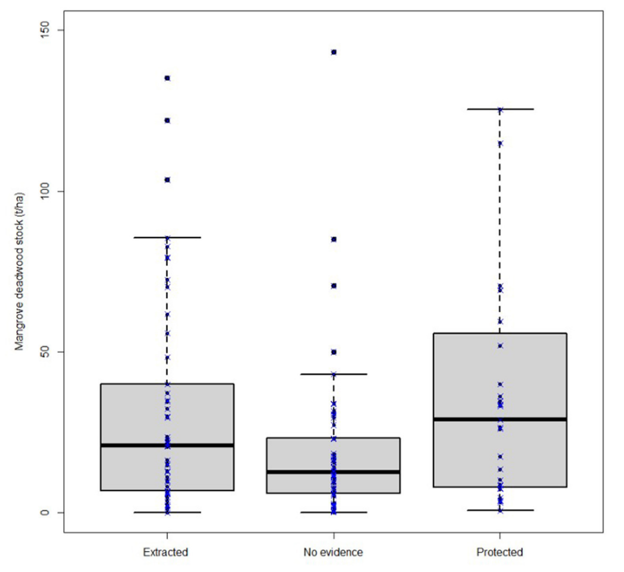

FIGURE 6 | Mangrove dead wood stocks Mg ha^{-1} under different reported human impacts based on available evidence from the literature review.

Statistical Synthesis

Stock of Dead Wood in Mangrove Forests

Standing stocks of DW were reported from sites in the Americas (7), Africa (6), Asia (4), the Caribbean (3), Oceania (3), and Middle East (1) (**Supplementary Table 1**). The mean (± SD) biomass from these sites was 16.85 ± 25.35 Mg ha^{-1} (median = 9.4, IQR: 1.98–18.27 Mg ha^{-1}). The largest stock of standing DW was 143.2 Mg ha^{-1}, reported in an *Avicennia germinas* dominated stand in French Guiana, while the lowest stocks

TABLE 2 | Mean ± SD mangrove dead wood production (Mg ha^{-1} yr^{-1}) from different estimation methods.

Method	No. of sites	Production estimate
Self-reporting	3	0.43 ± 0.49
Mortality rate	12	8.43 ± 9.60
Self-thinning/mortality model	2	2.40 ± 2.76

(<0.03 Mg ha^{-1}) were reported from mangroves with intensive human impact at Mombasa, Kenya and managed forests in Fiji (**Supplementary Table 1**). Relatively low values of standing DW were also reported in two stands in Australia: 0.78 Mg ha^{-1} in a young *Rhizophora* dominated forest in Missionary Bay, and 1.91 Mg ha^{-1} in Port Douglas.

Downed wood biomass was reported in 34 articles (76 sites); 12 of them studied Asian stands, 11 in the Americas, six in Oceania, four in Africa and one in the Caribbean (**Supplementary Table 1**). The mean biomass of downed wood was 29.92 ± 36.72 Mg ha^{-1} (median = 15.84, IQR: 7.40–34.00 Mg ha^{-1}); the highest (between 115.00 and 179.20 Mg ha^{-1}) were reported in *Rhizophoraceae spps* stands in Fiji. The lowest stock (0.26 Mg ha^{-1}) was recorded in an *A. marina* dominated stand in Sofala Bay, Mozambique where there was evidence of wood extraction for fuel, charcoal and building material.

There were nine cases in which total DW was reported, and it averaged (mean ± SD) 31.76 ± 24.68 Mg ha^{-1} (IQR: 12.65–44.18 Mg ha^{-1}). Six of the articles studied sites in Asia, while the others were from Africa (2) and the Americas (1). The highest stock of total mangrove DW, 85.40 Mg ha^{-1}, was reported in Bunaken National Park in Indonesia, while the lowest, 2.40 Mg ha^{-1}, was reported from Mexico.

Data on total aboveground mangrove DW (combining standing and downed) were collated and/or derived from values of downed wood, total and standing DW found in 57 articles (120 sites) and the additional nine sites in Kenya. The total stock of DW averaged (mean ± SD) 29.65 ± 35.32 Mg ha^{-1} (median = 16.40 IQR: 6.80–34.80 Mg ha^{-1}). The heavily exploited site of Mombasa, Kenya and a forest in the USA recorded the lowest values of DW stocks, 0.03 and 0.16 Mg ha^{-1}, respectively. The highest stocks were reported in a French Guiana stand (143.20 Mg ha^{-1}), and in Fiji with values between 135.20 Mg ha^{-1} and 180.4 Mg ha^{-1}. The lowest mean was reported from the Middle East (1.06 Mg ha^{-1}) in contrast to the Oceania where the widest range and the highest average were reported. The stocks of DW were significantly different between the regions; (**Figure 5**; Kruskal–Wallis test $\chi^2 = 20.53$, df = 5, $p < 0.001$). *Post-hoc* Dunn's test with Bonferroni adjustments showed stocks of DW were significantly higher in Asia and Oceania than Africa (0.002 and 0.026, respectively). No other differences were statistically significant.

The sites reporting adverse human impacts and extraction of mangrove wood had significantly lower stocks of DW (mean 32.08 ± 33.50; median = 20.90 IQR: 6.80–40.00 Mg ha^{-1}) as compared to those reported to be protected (mean = 41.66 ± 46.50; median = 28.95 IQR: 8.01–55.70 Mg ha^{-1}; **Figure 6**; Kruskal–Wallis test $\chi^2 = 6.22$, df = 2, $p = 0.045$).

Production of Dead Wood in Mangrove Forests

In our study, we found only three sites—in Australia and China—that explicitly reported the rate of production of DW in mangroves (**Supplementary Table 1**). In addition to these, data on mortality rates were used to estimate DW production at 12 sites and mortality/thinning models could be applied to two sites; there were no significant differences among medians from these three estimates (Kruskal–Wallis test) (**Table 2**). The spatial distribution of the sites was: Africa (3), Asia (7), Oceania (5), and Americas (2) (**Figure 4**), with the most recent of the studies in 2019 in Japan, and the oldest data reported (1986) for a site in Malaysia. Highest and lowest rates of mangrove DW production were 26.68 Mg ha^{-1} yr^{-1} for a *Bruguiera* species forest in Japan and 0.01 Mg ha^{-1} yr^{-1} for a young *Rhizophora* spp Australian stand, respectively.

Given the limited data and uncertainties over the underlying distribution, bootstrapping (10,000 iterations) was used to estimate an average and non-parametric 95% confidence intervals (using the BCa procedure) for these productivity data of 6.30 (3.10–11.40) Mg ha^{-1} yr^{-1}. Assuming a per capita consumption of 1.2 Kg day^{-1} (Jung and Huxham, 2018) these estimates suggest a community of 1,000 people would need a forest of between 38.4 and 141.3 ha in size to provide their fuelwood needs sustainably.

DISCUSSION

The current study estimates averages of mangrove above-ground DW stock (29.65 ± 35.32 Mg ha^{-1} (mean ± SD); based on information from 129 sites) and production (6. 30: 3.10–11.40 Mg ha^{-1} yr^{-1} (mean with 95% non-parametric CI); based on information from 16 sites) derived from a dataset with values from Africa, Americas, Caribbean, Oceania and Middle East. However, given the small number of relevant studies and patchy geographic coverage these estimates need to be treated as very provisional. For example regions, including West Africa and West America, are not well-represented in the literature, even though these areas constitute a considerable percentage of the total global mangrove coverage (Spalding et al., 2010). Hence a key finding of the current work is that mangrove DW is relatively under-researched and deserves further study.

Forest DW is an important part of the global carbon pool (IPCC, 2014). It accounts for around 8% (∼73 Pg) of all carbon in terrestrial forests (Pan et al., 2011). It may rival or exceed other major carbon pools in individual forests. For example, the carbon found in the DW pool in boreal forests represents almost twice (178%) that carbon found in the soil pool (with 27% and 43% for temperate and tropical forests respectively) (Pan et al., 2011). The data presented here suggest that per unit area stocks of DW in mangroves are within the range (although toward the bottom end) of those found in other forests. The IPCC gives median DW stocks of 18.2, 43.4, 34.7, and 10.7 Mg ha^{-1} for tropical, evergreen and deciduous forests (IPCC, 2003; Table 3.2.2), and for mature mangroves (IPCC, 2014; Table 4.7), respectively; although stocks in particular forests may far exceed these. There is evidence that DW is increasing in many forest types, both in total amounts and as a proportion of total pools, as degradation and disturbance spreads. Pan et al. (2011) report a "large sink increase" of deadwood in boreal forests over the decade up to 2007, caused by increasing climate-related disturbances and further suggest increased "dead biomass production" in tropical forests. Case studies of logging and other intense anthropogenic disturbance typically show increases in DW. For example, DW increased from 55 Mg ha^{-1} in intact Brazilian rainforest, to 75 Mg ha^{-1} with reduced-impact logging, to almost 110 Mg ha^{-1} in a logged forest (Keller et al., 2004). In a study of carbon stocks at different times following logging in Indonesian mangroves, Murdiyarso et al. (2021) found DW stocks immediately following logging that were double those in protected forests (39.73 vs. 19.98 Mg C ha $^{-1}$ respectively). Although stocks may be comparable, it is likely that above ground DW is not proportionally as important a pool in most mangroves as in most other forest types, principally because of the dominance of the soil carbon pool in mangroves, which often exceeds 90% of the total carbon present (Gress et al., 2017). Our estimates of mangrove DW do not include below ground data and hence underestimate total stocks and productivity. Including information on below ground DW may have major impacts on estimates of total stocks at some sites. For example, at Gazi Bay in Kenya, there are 32.5 Mg ha^{-1} of below ground dead roots in natural *Rhizophora spp* dominated stands (Tamooh et al., 2008), which exceeds the aboveground stocks, 0.62 Mg ha^{-1}, reported here. However, soil carbon is likely to remain the dominant pool at most sites even if below ground DW is included.

The wide variation in the estimates of DW reported here could represent a sparse sample from a large and variable population but might also imply use of dissimilar measurement/monitoring systems in mangrove DW stock and production assessments. The protocols for estimating mangrove DW stocks described (Kauffman and Donato, 2012) are now widely used and will help to address historical differences in methods; their adoption and refinement within IPCC guidelines means that inconsistency in methods does not appear to be a major problem in comparing DW stock estimates between sites. Instead, the spread in the values from different studies is probably related to a wide range of biological, geographical and anthropogenic factors, including the age and structural characteristics of the mangrove forests. For example, Robertson and Daniel (1989) reported that DW stock for a mature stand was 8-fold that of a young stand at a similar protected site in Australia (14.89 and 1.81 Mg ha^{-1}, respectively). Terrestrial stands have similarly been found to contain very variable amounts of DW controlled by natural forest dynamics and human impact (Harmon et al., 1986; Sandström et al., 2019). Meta-analytical examination of the drivers of this variance (between, for example, forests of different ages, species and geomorphological settings) is an obvious research goal but was not possible here given the paucity of data. In contrast to the measurement of stocks, there are no standard methodological approaches for establishing mangrove DW production rates (which is a much harder variable to estimate, but which is more relevant for consideration of fuelwood extraction and management and also contributes to our understanding of mangroves as a carbon sink). Further theoretical and empirical work on DW productivity in mangroves is highlighted by this review as an important target for future research.

One motivation for the current work, and justification for a focus on above ground DW, was to contribute toward understanding and managing human needs for dead wood from mangroves (which is principally for fuelwood). The data presented here (that excludes cases of instantaneous high mortality) show that mangrove forests exposed to human exploitation are generally lower in total and standing DW stocks than those that are relatively undisturbed. Logic, along with multiple site-specific studies looking at human impact on mangroves (for example (Huxham et al., 2017; Chow, 2018; Adanguidi et al., 2020) and relevant reviews (e.g., Chowdhury et al., 2017; Huxham et al., 2017), suggest that extraction of fuelwood will lower stocks of DW and when intense could have a range of impacts on forest ecology; if fuelwood collection includes the cutting of living biomass then it may quickly threaten the sustainability of a forest. Given the importance of this ecosystem service for the lives of millions of people, along with the implications of excessive fuelwood collection for mangrove conservation, it is surprising that so little attention has been given to it, from either scientific or management perspectives. There are examples of the sustainable management of fuelwood and charcoal extraction from mangroves. For example McNally et al. (2011) describe how management of mangroves in the Saadani National Park, Tanzania, has reduced unsustainable cutting, leading to a 5% reduction in households using mangrove fuelwood. This was combined with an overall increase in income from more productive shrimp capture fisheries. Importantly, the reduction in fuelwood use was recorded predominantly in richer households, which could afford to shift to alternative sources of fuel, so this example appears to show how mangrove conservation can lead to enhanced economic opportunities for local communities, without penalizing the poor and whilst still permitting the use of fuelwood by those who need it most. A very different model of sustainable use comes from the Matang mangrove in Malaysia, which has been producing commercial charcoal for more than a century (Goessens et al., 2014). In addition, communities around the world have used mangrove fuelwood sustainably for a long time (Bosire et al., 2015), applying a wide range of customary management practices. Hence, we can learn from these examples to help manage fuelwood at other sites; doing so will usually require much better information on levels and productivity of DW than is available at present.

CONCLUSION

The present study adds to knowledge on mangrove DW stocks and production and reveals the current paucity of information and research on these topics, both globally and particularly for some major regions.

The data presented has evidenced lower DW stocks in mangroves compared to terrestrial counterparts, although the values available fall within the very large range reported in other tropical forests. Research on terrestrial forests has shown the influence of latitude, age and species on DW stocks and production. The importance of such drivers in mangroves remains unknown and should be the subject of future research. More studies within and across regions are necessary to make clear patterns and to quantify the ecological roles and thresholds of DW stocks and production in mangrove forests. Whereas, simple methods for quantifying stocks are available and widely used, estimation of DW production in mangroves remains a challenge and has rarely been attempted. Lastly, the importance of mangrove fuelwood to millions of people should encourage further work on understanding production and on using this understanding to assist with sustainable management of this valuable resource.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author/s.

AUTHOR CONTRIBUTIONS

MH, LM, and DK were involved in the conception and development of the presented review. LM and DK performed the literature search under the supervision of MH. JK, MH, and LM were involved in the conceptualization, supervision, collection, and analysis of the data from the Kenya sites. LM performed the analysis for the presented review and together with MH designed figures. DK developed the images. LM took lead in writing the manuscript in consultation with MH. All authors provided critical feedback on the manuscript.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/ffgc.2022.767337/full#supplementary-material

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APPENDIX

Appendix 1. Research areas excluded during the "Web of Science" search.

"*Mathematical computational biology; History; Endocrinology metabolism; Cardiovascular system, Cardiology; Evolutionary biology; Imaging science photographic technology; Infectious diseases; Communication; Materials science; Demography; Behavioral sciences; Health care sciences, services; Pathology; Polymer science; Geochemistry, geophysics; Education; Educational research; Public administration; Oncology; Mycology; Life sciences, biomedicine, other topics; Instruments, instrumentation; Sport sciences; Science technology, other topics; Automation control systems; Physiology; Psychology; Microscopy; Physical geography; Neurosciences, neurology; Cell biology; Women's studies; Entomology; Dermatology; Nutrition dietetics; Veterinary sciences; Integrative complementary medicine; Microbiology; Immunology; Spectroscopy; Developmental biology; Anthropology; Urban studies; Biotechnology, applied microbiology; Paleontology; Public environmental occupational health; Government law; Electrochemistry; Reproductive biology; Biophysics; Mining, mineral processing; Genetics heredity; Optics; Chemistry; Parasitology; Radiology, nuclear medicine, medical imaging; Geology; Sociology; General internal medicine; Anatomy morphology; physics; History, philosophy of science; Pharmacology, pharmacy; Operations Research, management science; Toxicology; Information science, library science; Research experimental medicine; Archaeology; Telecommunications; Physical sciences, other topics; International relations; Tropical medicine; Engineering; Gastroenterology, hepatology; Area studies; Hematology; Construction building technology; Mathematics; Respiratory system; Geriatrics gerontology; Food science technology; Computer science; Remote sensing; Biochemistry, molecular biology; Architecture; Art; Zoology; Business economics*."

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Quantifying the Reporting, Coverage and Consistency of Key Indicators in Mangrove Restoration Projects

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Mangroves are often cleared for aquaculture, agriculture, and coastal development despite the range of benefits for people and nature that they provide. In response to these losses, there are multiple global, and regional efforts aimed at accelerating mangrove forest restoration, resulting in many restoration projects being implemented and managed by different groups with highly diverse objectives. The information reported from these restoration projects is extremely variable, limiting our ability to identify whether desired objectives have been met or key factors that determine effective and durable restoration have been applied. To address this problem, we developed a holistic monitoring framework that captures the key indicators of restoration, spanning project aims, intervention type, costs, and ecological and socioeconomic outcomes. Subsequently, using a systematic literature search, we examined 123 published case studies to identify the range and quality of reported information on restoration, relative to our framework. We found that there were many gaps in reporting, for multiple indicators. Sections related to site conditions prior to restoration (reported in only 32% of case studies) and socioeconomic outcomes (26%) were consistently missing from most project reporting. Conversely, information on the type of intervention was reported for all case studies, and the aims of the project (reported in 76% of case studies) and ecological monitoring (82%) were far more prevalent. Generally, the restoration literature did not follow any specific framework in terms of reporting which likely contributed to the gaps in the information recorded. These gaps hinder comparisons between case studies, inhibiting the ability to learn lessons from previous restoration attempts by identifying commonalities. The need for more structure and consistent reporting supports the development of a standard restoration tracking tool that can facilitate the comparison of restoration efforts, aiding the implementation of future projects.

Keywords: mangrove, restoration, framework, taxonomy, evidence-based practice, monitoring

INTRODUCTION

Mangroves support a wide variety of ecosystem processes, functions, and services (Dahdouh-Guebas et al., 2021). These range from local-scale benefits such as timber (Palacios and Cantera, 2017), fishery enhancement (Carrasquilla-Henao and Juanes, 2017), protection against storm events (Marois and Mitsch, 2015), and mangrove-associated tourism (Spalding and Parrett, 2019), to global-scale benefits such as carbon sequestration (Donato et al., 2011; Richards et al., 2020). Mangrove forests also support a wide variety of marine and terrestrial species, including plant species and marine megafauna of conservation concern (Polidoro et al., 2010; Sievers et al., 2019). Despite these benefits, between 1980 and 2005 an estimated 20% of global mangrove forests were lost (FAO, 2007). Mangrove forest loss has largely been attributed to conversion to aquaculture and agriculture (Richards and Friess, 2016; Thomas et al., 2017; Goldberg et al., 2020) and chronic overexploitation (Ilman et al., 2016). To combat these losses, there have been many recent efforts to restore mangrove forests (Lee et al., 2019); however, many of these have failed to establish natural functioning mangrove ecosystems (Barnuevo et al., 2017).

Mangrove restoration involves a range of interventions from restoring hydrological connectivity and increasing sediment capture, to natural regeneration, but the dominant strategy globally has been mangrove planting, often of a single species from a limited number of families (Kairo et al., 2001; López-Portillo et al., 2017; Lee et al., 2019). This approach has become widespread because of its relatively low costs and reporting indicators focused on number of seedlings planted (Bayraktarov et al., 2016). However, post-planting survival is typically low (Saenger and Siddiqi, 1993; Primavera and Esteban, 2008; Samson and Rollon, 2008), often due to the planting of incorrect species in locations that are unsuitable in terms of hydrology and salinity for mangrove forests to establish (Elster, 2000; Kodikara et al., 2017; Wodehouse and Rayment, 2019). Failures in mangrove restoration are also compounded by challenges associated with the socio-economic and political landscape of the area being restored (Gallup et al., 2020), with the inability of many restoration projects to adequately address governance issues often cited. Successful management of mangrove areas, which are complex socio-economic systems, requires a clear understanding of the needs of different stakeholders and the inclusion of local people in the decision making process (Hugé et al., 2016; Frank et al., 2017; Vande Velde et al., 2019; Martínez-Espinosa et al., 2020). However, practitioners are frequently working to short delivery timeframes from funding agencies, resulting in interventions occurring on land where land tenure is less contested, for example mudflats and seagrass meadows. These areas are usually unsuitable in terms of the physiological tolerances of the planted species (Lovelock and Brown, 2019) and can cause damage to these other coastal ecosystems. Despite these failures, there are also many examples of successful mangrove restoration projects (Saunders et al., 2020). For example, approaches have been implemented that involve local community participation and include a focus on non-planting restoration activities (Brown et al., 2014; Zaldívar-Jiménez et al., 2017). While planting is often cited as having low long term survival in some instances through altering local hydrodynamics and physicochemical conditions, mangrove planting may facilitate natural regeneration of other mangrove species (Bosire et al., 2003), encourage faunal recolonization (Bosire et al., 2004, 2008; Walton et al., 2006; Canales-Delgadillo et al., 2019), and provide the basis for the development toward a naturally functioning mangrove forest (Bosire et al., 2006, 2008; Tamooh et al., 2008).

The increase in mangrove restoration effort is underpinned by a number of global initiatives, such as the UN Decade of Restoration (Waltham et al., 2020), the Bonn Challenge,[1] country-level climate commitments (such as Nationally Determined Contributions as part of the Paris Agreement), and global conservation partnerships, such as the Global Mangrove Alliance.[2] Further encouragement for restoration is being given by both growing social and economic arguments for the benefits of mangrove restoration, and new work to help identify optional locations for restoration (Worthington and Spalding, 2018). Despite the large economic investments this will entail, it is still unclear whether global restoration targets can, or will, be met. Information on mangrove restoration efforts is disparate, with project outcomes often reported in gray literature (if documented at all), and project failures often underreported. Even where information is available, outcome indicators are inconsistent between projects, limiting our ability to learn from these projects to improve future restoration.

This study aims to improve our understanding of the problem of highly variable approaches and inconsistent outcomes from mangrove restoration efforts, and of a failure to disseminate and share lessons. While there have been a number of reviews looking at certain aspects of mangrove restoration project implementation such as costs (Bayraktarov et al., 2016), motivations and outcomes (Bayraktarov et al., 2020; Cadier et al., 2020; Su et al., 2021), here we develop a framework of key metrics and indicators that would enable a holistic description of any restoration attempt. The framework identifies the full range of factors that should be considered when planning, implementing and monitoring a mangrove restoration project and we apply this framework to peer-reviewed mangrove restoration literature to determine current reporting coverage. By identifying what is, and is not, being recorded, we can quantify knowledge gaps and highlight opportunities and benefits of more comprehensive and consistent reporting of indicators and outcomes.

METHODOLOGY

To determine the detail and consistency of reporting on mangrove restoration projects, their approaches, and outcomes, a review of the primary literature was undertaken. The work consisted of two broad approaches: the development of an idealized Candidate Indicator Set (CIS) of metrics and indicators that would be required to comprehensively report on a mangrove

[1] https://www.bonnchallenge.org/

[2] https://www.mangrovealliance.org

restoration project; and a comprehensive review of the primary literature describing restoration efforts world-wide.

Candidate Indicator Set

The CIS was developed to capture the key aspects of mangrove restoration projects. The initial structure of the CIS was based on a previous preliminary synthesis of mangrove restoration projects (Worthington and Spalding, 2018) and was further developed after reviewing key mangrove restoration literature and discussions amongst the authors. The framework was divided into 10 sections, with each section addressing a different aspect of a mangrove restoration project. Within each section there were several indicators for which data could be recorded. Where possible, the potential responses to an indicator were in the form of predefined categories, although freeform answers were allowed if the information did not fit with one of our pre-determined groupings (see **Supplementary Material 1**). For example, the type of project could be "Restoration," "Rehabilitation," "Protection," "Bioremediation" or a freeform "Other." The CIS attempts to capture all the salient information that would be required to comprehensively describe and monitor a mangrove restoration project (**Table 1**).

Searches

To assess coverage of the CIS we conducted a literature review of scientific articles that described mangrove restoration and/or rehabilitation case studies. The identification of relevant mangrove restoration case studies (**Figure 1**) followed the Reporting standards for Systematic Evidence Syntheses methodology (Haddaway et al., 2018). The searches were limited to abstracts, titles, author keywords and keywords plus in Web of Science (WoS) (coverage 1970—to date),[3] and abstract, titles, and keywords in Scopus (coverage 1788—to date).[4] The searches were limited to English language studies. To identify the literature the following search string was used: (mangrove OR mangal) and (restor* OR afforest OR rehab* or planting). Searches were run on 4th February 2020. It should be noted that our search string will not capture all mangrove restoration

[3] www.webofknowledge.com

[4] www.scopus.com

TABLE 1 | Overview of the mangrove restoration assessment framework.

Section	Description	Example information recorded
Project details	Background information on the restoration project including the type of project (e.g., restoration, afforestation, rehabilitation, bioremediation, or protection), its location and the land tenure and management status of the site.	Project title Type of project Site location Duration of project Management status Land tenure
Project cost	Whether the project had funding and its sources. The costs of the project are captured following Spurgeon (1999) and Iacona et al. (2018), where possible splitting the expenditure between capital, construction, and operational costs. The organizations participating and managing the project were also recorded.	Project funding Project initiated by Project lead Partner organizations Capital costs Operational costs Other costs
Project aims	The aim(s) of the restoration project.	Project aims
Causes of decline	The causes of the decline in the site's mangroves based on the IUCN Threats Classification Scheme (IUCN, 2021).	Causes of decline Consequences of decline
Site conditions	The pre-intervention activities and the underlying site conditions.	Site assessment Pilot study Expert consultation
Physical interventions	Type of restoration intervention applied, details on species and source of materials for planting projects.	Size of project area Methods used Species planted
Community	Awareness programs and community leadership or engagement. Post-restoration management and land tenure.	Awareness/involvement activities and training programs Volunteer/community participation Management of area Regulatory/protection Regime in place
Monitoring	Description of the post-intervention ecological monitoring including its duration and the organization involved.	Duration of monitoring Monitoring conducted by
Ecological outcomes	Metrics of ecological success, including the presence of natural recruitment and seedling survival.	Percentage seedling Survival Natural recruitment Goals of restoration met
Socioeconomic outcomes	Metrics of socioeconomic success assessed using the framework developed by McKinnon et al. (2016).	Socioeconomic outcomes

*See **Supplementary Material 1** for full framework.*

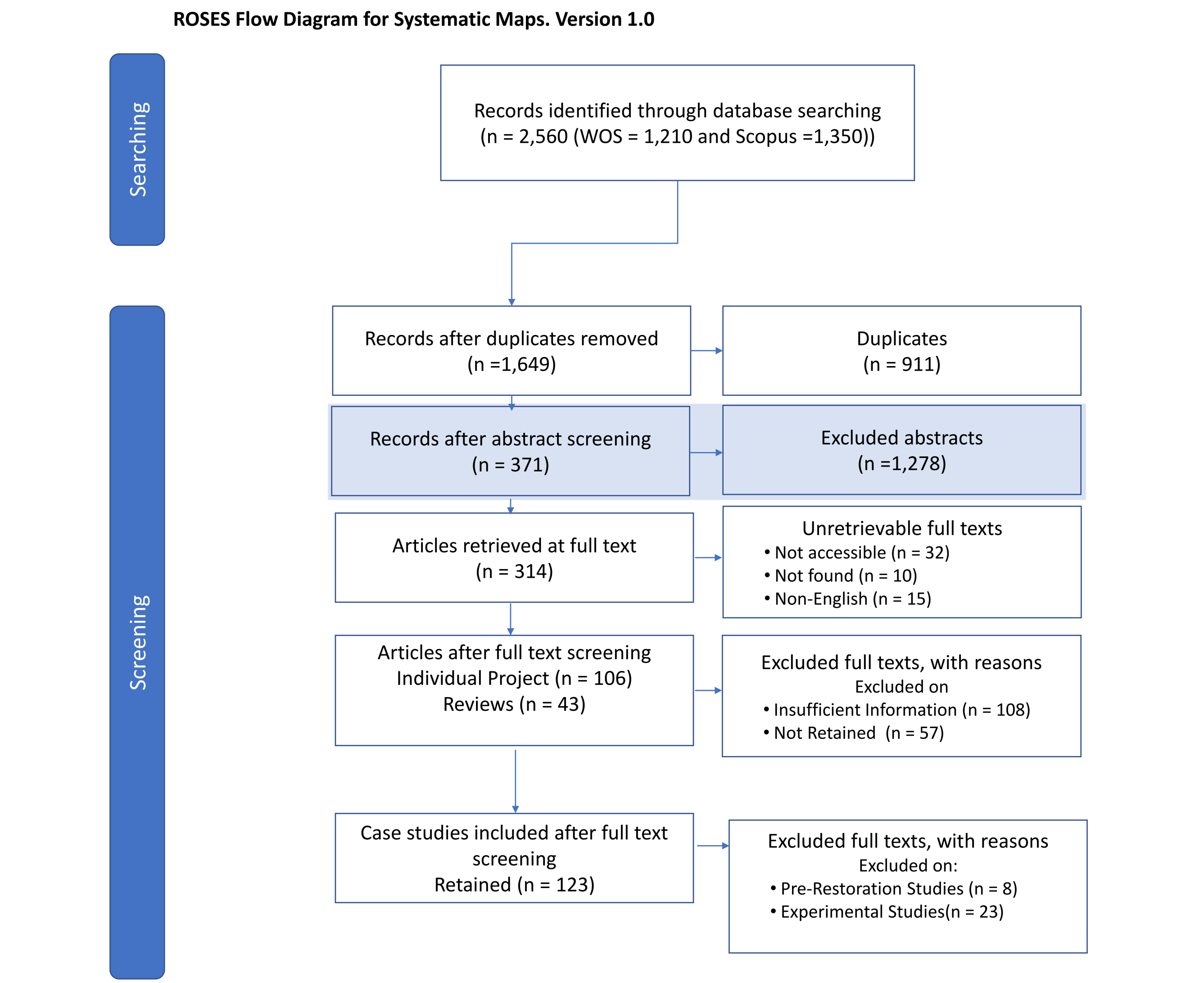

FIGURE 1 | ROSES flow diagram of the literature breakdown through screening.

case studies as such projects have been described using a variety of other terms including "replanting," "reafforestation" or "plantation" (López-Portillo et al., 2017). However, we believe our systematic approach results in a dataset indicative of the prevailing trends in restoration project reporting.

Screening

In total 2,560 articles (WoS = 1,210 and Scopus = 1,350) (**Figure 1**) were identified by the search string. Duplicates between the two databases were manually removed, resulting in 1,649 articles. A benchmark test was conducted to quantify how well our search string captured the published literature on mangrove restoration projects. The database of mangrove restoration project literature used in the benchmark test was obtained from a previous coastal restoration synthesis (Bayraktarov et al., 2016). Out of the 54 mangrove publications cited in Bayraktarov et al. (2016), 26 were journal articles that should have been captured by our search of WoS and Scopus (the remaining entries were webpages, book chapters, and reports that are less likely to be within the WoS and Scopus databases). Of these 26 journal articles, our search found 22, with the four missing articles comprising two articles from journals not indexed in WoS or Scopus, one from a newsletter not indexed in WoS or Scopus, and one article that, while indexed, did not include any of our search string terms in the abstract or title.

Given that the benchmark test suggested our search gave good coverage of the literature, we screened the 1,649 search results for restoration projects, using both titles, and abstracts for initial inclusion/exclusion decisions. We developed an inclusion/exclusion criterion for reviewing titles and abstracts following the SPIDER (Sample, Phenomenon of Interest, Design, Evaluation, Research type) protocol to ensure consistency in decisions (Cooke et al., 2012). To test the accuracy and repeatability of our inclusion procedure a kappa test was undertaken, 10% of the abstracts were chosen randomly and labeled by two reviewers (YMG; TAW) as either being a restoration project or not. The kappa statistic for this 10% of abstracts was 0.26 suggesting only a "fair" level of agreement between the two reviewers (Landis and Koch, 1977). Therefore, the inclusion criteria were revisited, with the aim of being more inclusive when determining whether to retain a study for the database based on the title and abstract (**Table 2**). As such, in the revised criteria a paper was retained if the abstract or title only briefly mentioned a restoration project, the restoration case study itself did not have to be the focus of the abstract. A second random 10% sample of the articles was chosen and assessed by the same two reviewers. The more inclusive criteria resulted in a "substantial agreement" (kappa statistic = 0.77) (Landis and Koch, 1977).

After obtaining a high kappa value the remaining 80% of the abstracts were screened by a single reviewer (YMG) using the new inclusion criteria. In total, 371 abstracts were found that potentially contained restoration projects (**Figure 1**). Full texts were sought for the 371 articles; however, 10 were not found, 32 were not accessible and 15 were not in English. The full texts of the remaining 314 journal articles were reviewed to determine whether they contained information on a single or multiple mangrove restoration project(s) (relevant information, n = 257), or not (no relevant information, n = 57). These 257 papers were then placed into one of three categories based on the amount of information on restoration projects they contained: "individual project," "review," or "insufficient information." For a paper to be labeled as an "individual project" (n = 106), it required at least three of the following key pieces of information: the project location, the duration of the physical restoration activities (project duration), the restoration method used, the duration of post-restoration monitoring and results that encompassed any form of monitoring data that was recorded post restoration (**Table 3**). We chose these key pieces of information as indicators as to whether to retain a study as we assumed them to be the most regularly recorded types of information for mangrove restoration projects. By targeting papers that had a minimum amount information, we aimed to ensure that sufficient data could be extracted from the paper to contribute to the review. Reviews (n = 43) were individual articles that contained data from multiple projects meeting the individual project criteria. Many papers reviewed at the full text stage cited a restoration project had been undertaken but contained little information except its approximate location, these were labeled as insufficient information papers (n = 108). Given that the focus of these papers was generally not to describe in detail a restoration project but rather provide it as an example of restoration in the context of a scientific piece of research, we deemed that they would not contribute sufficient data to the review, and they were therefore not retained. Papers marked as an "individual project," or a "review" were retained (n = 149). A random 10% of the full text papers were chosen and labeled by two reviewers (YMG; TAW) as either not relevant or relevant with the kappa statistic between the two reviewers 0.83 considered as "substantial" in terms of strength of agreement (Landis and Koch, 1977). The same 10% of papers were also labeled by the two reviewers as: individual projects, reviews and insufficient information. The kappa statistic was 0.71, subsequently the remaining 90% of papers were screened by one reviewer.

TABLE 2 | Spider protocol applied for abstract inclusion/exclusion (Cooke et al., 2012).

S	PI	D	E	R
Sample	**Phenomenon of interest**	**Design**	**Evaluation**	**Research type**
1,649 Journal abstracts and titles	Mangrove restoration field studies	Abstracts reviewed and retained or rejected	Abstracts retained at any mention of a restoration case study	Mixed

A final assessment of the 149 "individual project" and "review" papers was undertaken to identify case studies for which data could be extracted. Case studies are defined as a unique individual restoration project that have data available sourced from both the "individual project" and "review" papers. At this stage, the 106 "individual project" papers were further subdivided into three groups: "pre-restoration studies", "experimental studies" and "case studies". Pre-restoration studies (n = 8) outlined initial intervention assessments but did not contain information on an actual restoration effort. Experimental studies (n = 23) were methodological papers on specific aspects of mangrove restoration (e.g., statistical comparisons on different planting approaches) but lacked wider framing of mangrove restoration as a conservation intervention (e.g., absence of non-research project aims, socio-economic setting or outcomes of the restoration effort). Once the "pre-restoration studies" and "experimental studies" were removed, this resulted in 75 "case studies" for data extraction.

The "review" papers were then assessed to identify additional "case studies." As the review journal articles contained multiple projects, each project was identified and any in-text references describing the project were collated. If, for a project, the information in the review and the associated references contained at least three key pieces of information (see above) it was retained for data extraction. As "review" papers often contained information on the same restoration projects, only 48 "case studies" were identified from the 43 papers. The final dataset comprised 123 restoration case studies (see **Supplementary Material 2**).

Recording the Data

The CIS framework was then used to extract data from the 123 case studies. An initial consistency assessment was carried out using five randomly selected case studies, which were reviewed by two reviewers (YMG; TAW). The results from the two reviewers were compared, with similar responses for the majority of categories. Where there were disparities in what was recorded between the reviewers, this was discussed and the remaining 118 case studies were screened by a single reviewer. The case study papers occasionally had in-text references to papers that held more information about the restoration case study under

TABLE 3 | SPIDER protocol applied for inclusion/exclusion based on article full text (Cooke et al., 2012).

S	PI	D	E	R
Sample	**Phenomenon of interest**	**Design**	**Evaluation**	**Research type**
314 Journal articles	Mangrove restoration projects	Papers defined as: (1) Relevant or not relevant (2) Retained or not retained	Papers categorzed as: (1a) Not relevant—no information on a mangrove restoration project (1b) Relevant—information on a mangrove restoration project (2a) Retained—when an article contained three or more key pieces of information for an "individual project" or "review" (2b) Not retained—contained fewer than three of the key pieces of information labeled as "insufficient information"	Mixed

consideration. In these cases, we searched for the referenced papers and, if found, relevant information was also extracted from this source. The final 123 case studies are not an exhaustive list of mangrove restoration projects referred to in the published literature. For instance we did not capture all the projects identified in Bayraktarov et al. (2016) and López-Portillo et al. (2017). This is due to several factors (1) our search string did not capture all the terms that have been used to describe mangrove restoration (see above) (2) our searches were confined to English language studies biasing against regions where English is less widely used (3) we only included studies from the published literature which is unlikely to be the medium of publication for many organizations and (4) our inclusion criteria removed studies that only provided a very limited description of a restoration project. However, our approach to surveying the literature was systematic and we believe the results are indicative of the overall trends in reporting, coverage, and consistency of key indicators in mangrove restoration projects.

RESULTS AND DISCUSSION

Articles with case study information were published between 1990 and 2019, with restoration interventions commencing between 1957 and 2015. No case studies contained information for all the 10 CIS sections. Pre-restoration activities and socioeconomic outcomes were particularly data deficient with only 15% of case studies covering both of these categories (**Figure 2**). Conversely, information on the intervention itself and the cause of mangrove decline were recorded for 68% of case studies (**Figure 2**). Whilst our analysis suggests that 80% of the case studies had information on costs, it should be noted that these papers mainly referred to the funding source rather than providing a breakdown of the costs of the restoration itself.

Project Details

Restoration case studies were recorded in 24 countries with over three quarters carried out in Asia (**Figure 3**). This is unsurprising given that this region contains a large proportion of the global mangrove extent and has exhibited the greatest recent losses in mangrove area (Hamilton and Casey, 2016). However, this number is skewed by 23 case studies being recorded in Sri Lanka in a single review (Kodikara et al., 2017).

Only 12% of the case studies were recorded from Africa, the Middle East, the Caribbean or Central America, mirroring the broad scale patterns identified in a synthesis of marine and coastal restoration research (Bayraktarov et al., 2020). West and Central Africa only had 2% of the recorded case studies, despite the region supporting 14.5% of the global mangrove area. This could indicate that either few restoration attempts have been undertaken, that restoration has been undertaken but has not been recorded in the primary literature, or that limiting our search to English articles and removing non-English full-texts resulted in bias against regions where English is less

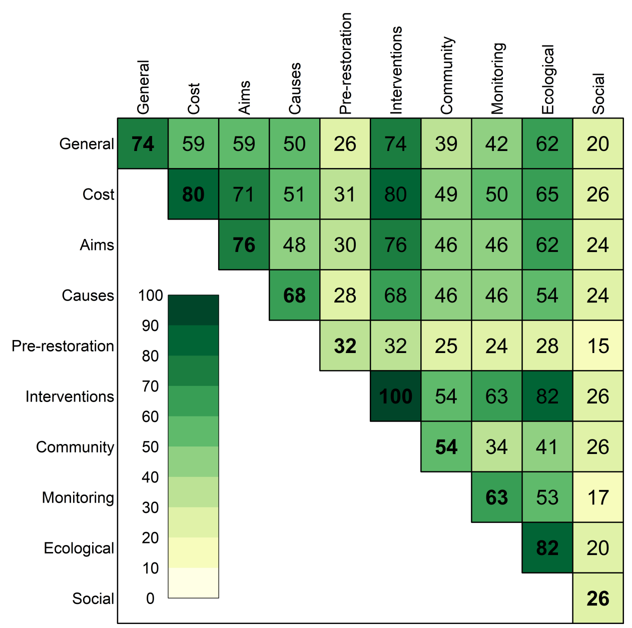

FIGURE 2 | Frequency of reporting across the different sections of our Candidate Indicator Set (CIS). A case study is considered to report on a section of the CIS if it contains any information addressing one or more indicators within the respective section, with the exception of the general section where only information for the indicators start and end dates, management status, and land tenure were used. The diagonal boxes (bold numbers) identify the percentage of case studies that record information for each single section of the CIS. The off-diagonal boxes represent the percentage of case studies that record information for the two interacting sections of the CIS.

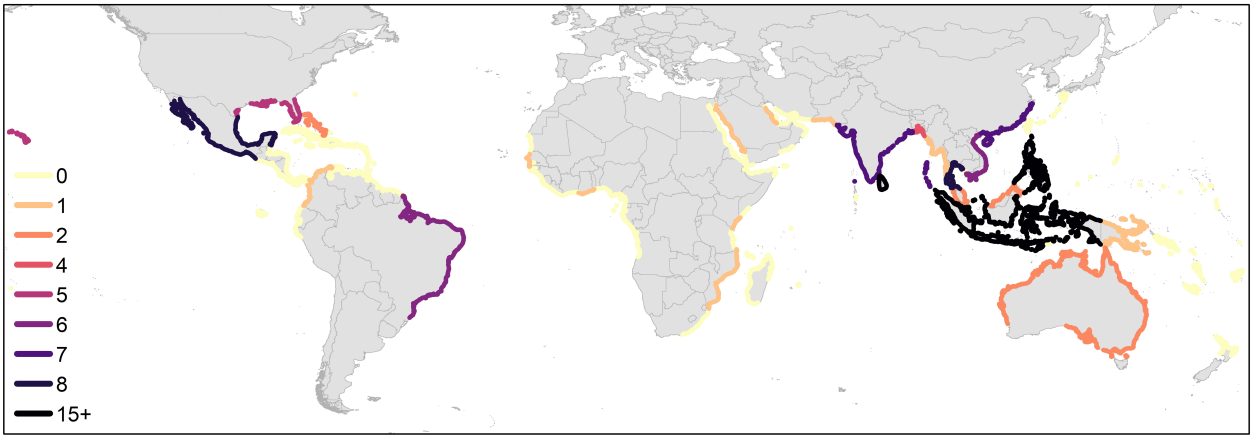

FIGURE 3 | Global map showing the location of countries that have mangroves and the number of restoration case studies recorded from each country.

widely used. Individuals in certain countries encounter barriers such as wealth, language biases, and security and geographical location challenges that reduce opportunities to collect or publish scientific data (Amano and Sutherland, 2013). These barriers are underscored by the observation that in terrestrial ecology and conservation studies Africa is highly underrepresented (Martin et al., 2012; Christie et al., 2020, 2021).

For the remaining project details' questions, very little information was available from our case studies. Management status, which recorded whether the site was protected as defined by the International Union for Conservation of Nature or Other Effective Area-Based Conservation Measure categories (Dudley, 2013; IUCN-WCPA Task Force on OECMs, 2019), was only recorded for 13% of case studies, with these few locations distributed across a range of management status designations. In addition, only 11% of case studies provide details on land tenure. Of the 13 studies that did record land tenure, six stated that the restoration sites were on land owned by the national government, with three each on communal and private lands. Given that the failure to understand and address potential challenges associated with land tenure has been highlighted as a key driver of restoration failure, because it influences the choice of restoration site and the continued resource use of an area (Mukherjee et al., 2015; Asante et al., 2017; Lovelock and Brown, 2019), this shortfall in reporting is a particular concern.

Project Costs

From the 123 case studies, 47 recorded their source of funding, 73 did not record if the project was funded, while just three cases stated that they were not funded. Over a third of case studies reporting a funding source had more than one (two funding sources $n = 11$, three $n = 5$ and six $n = 1$). For those case studies that explicitly stated the source of funding, there was an approximate 50:50 split between domestic and foreign funding sources. The most commonly cited source of funding was from the government or its agencies in the country where the restoration took place ($n = 22$), with foreign and international development banks ($n = 11$) and foreign governments and development agencies ($n = 9$; **Figure 4A**) also identified regularly. This is in line with a previous synthesis of marine and coastal restoration which highlighted government funding as the predominate source of mangrove, and other coastal ecosystem, funding (Bayraktarov et al., 2016).

While the source of funding was recorded for almost 40% of the 123 case studies, the total monetary value of that funding was only recorded in 14. Further, only eight case studies documented a breakdown of the costs enabling finer details of the expenditure, with the cost recorded varying hugely between case studies. The lack of reporting on project costs is an issue across conservation (Cook et al., 2017) and has stimulated a number of attempts to unify cost reporting frameworks (Iacona et al., 2018), such as those used here. This issue has been highlighted for mangrove restoration previously, with only 11% of the papers from a synthesis of marine restoration projects reporting project costs and a breakdown of expenditure (Bayraktarov et al., 2016). Due to this ongoing poor cost reporting, the ability to compare studies and evaluate the factors that modify costs and benefits is seriously hindered (Cook et al., 2017).

Project Aims

Across the 123 case studies, the majority (76%) stated at least one aim for the restoration, with many case studies having more than one (two aims, $n = 17$; three aims, $n = 6$ and four aims = 1). The most frequently recorded aims were related to coastal defense ($n = 33$), and biodiversity enhancement ($n = 31$, **Figure 4B**), which included a variety of different goals associated with increasing the area or functioning of the mangrove forests themselves. The prevalence of case studies whose aim was coastal defense was driven by the review of Kodikara et al. (2017) who examined mangrove planting following the 2004 Indian Ocean Tsunami. In addition, employment and income generation ($n = 9$), and enhancing shoreline stability and climate change resilience ($n = 12$) were frequently cited. It is notable that the stated aims across case studies were diverse, likely driven by the multitude of ecosystem services that mangroves provide (Brander et al., 2012).

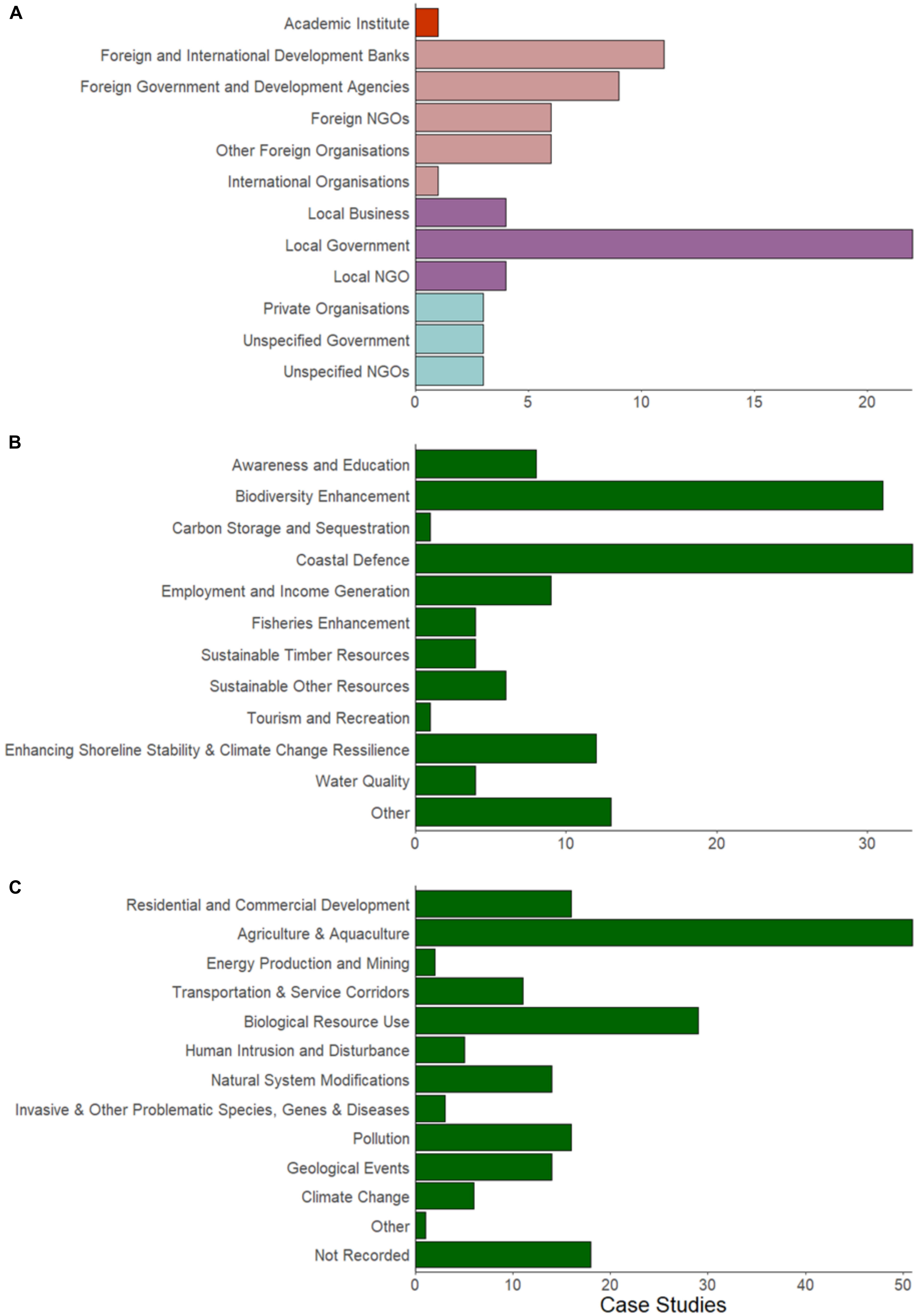

FIGURE 4 | (A) Source of funding recorded from the restoration case studies. The orange bar indicates funding from academic institutes, pink bars indicate funding from a foreign entity, purple bars indicate funding from local entities, light green bars indicate funding from unnamed governments NGOs and organizations. A case study could report multiple sources of funding. **(B)** Different project aims recorded in the restoration case studies. If a restoration project had more than one aim, all were recorded. **(C)** Causes of decline recorded in the restoration case studies. A case study could record multiple causes of decline.

Cause of Decline

The cause of the decline was recorded for the majority (67%) of the 123 case studies, with anthropogenic impacts rather than natural factors (e.g., erosion) predominantly identified. Over half of the 82 case studies that stated the cause of decline identified more than one (two causes of decline, n = 18, three, n = 20, four, n = 4 or five, n = 4), suggesting that mangroves are often subject to multifaceted human impacts. Within the IUCN Threats Classification Scheme (IUCN, 2021) the Level 1 categories "agriculture and aquaculture" and "biological resource use" (**Figure 4C**) were most frequently identified. Within "agriculture and aquaculture," aquaculture was the most regularly cited cause of mangrove decline and loss (Level 2 categories: aquaculture n = 40; agricultural expansion n = 11). After aquaculture, the second most common Level 2 cause of decline was wood harvesting (n = 27).

These results reflect the fact that the major driver of mangrove losses at a global scale is conversion to aquaculture and agriculture (Goldberg et al., 2020). The role of aquaculture has been particularly apparent in Southeast Asia (Richards and Friess, 2016; Thomas et al., 2017), where there was an industrial shrimp aquaculture boom in the 1980s and 1990s (Hall, 2003). Regarding wood harvesting, mangrove wood can provide important fuel and timber resource and as such chronic overharvesting has been identified as an issue in some areas (Iftekhar and Islam, 2004). Our results also identified 32 cases studies that identified urbanization or development (under the Level 1 categories "residential and commercial development," "transportation and service corridors" and "human intrusion and disturbance") as a major source of decline—in line with the findings of Dale et al. (2014).

Site Conditions Prior to Restoration

Twenty percent of the 123 case studies stated that some form of pre-intervention monitoring was carried out, with a further five specifying it hadn't been undertaken; however, for the majority (n = 93) this information was not recorded. Of the pre-restoration data available the most commonly recorded metrics were those related to the species present (n = 9 case studies), the geomorphic typology of the restoration area (n = 8) and the water or soil salinity (n = 7). In addition, six case studies specifically stated that they conducted a pilot restoration study. It is unknown whether these data are being collected for a greater proportion of restoration efforts and this information is just not presented in the published literature. However, this lack of information provides challenges for future restoration efforts as (1) the absence of details of initial conditions means before-and-after comparisons assessing intervention effectiveness are not possible and (2) knowledge of the pre-restoration biophysical setting may aid practitioners in identifying the most appropriate intervention.

Physical Interventions

The intervention method was stated in all except one of the case studies, with the most commonly used intervention being mangrove planting (n = 108), followed by hydrological restoration (n = 27; **Figure 5**). Approaches aimed at restoring hydraulic conditions (for example interventions such as fences to trap sediment and reduce erosion, n = 7) or altering site topography (such as restabilising the original site elevation, n = 4) were much less frequently recorded. There were also relatively few case studies that combined multiple restoration approaches, with only a combination of planting and hydrological restoration (n = 14) recorded in more than five case studies (**Figure 5A**). Hydrological restoration is more expensive than planting (Lewis, 2005) due to the need for heavy machinery and different countries have vastly different funding potentials (Bayraktarov et al., 2016). In addition, costs vary with location. For example, the median costs of mangrove restoration in the United States were recorded at US$100,861 ha^{-1} y^{-1}, compared to US$989 ha^{-1} y^{-1} for southeast Asian countries (Taillardat et al., 2020). Given that the majority of the case studies we reviewed occurred in Southeast Asia this may partly explain the high number of planting-based restoration methods observed. The timing of when the different intervention types were applied showed some variation. Within our case studies, pre-1990 only planting based restoration projects were recorded (**Figure 5B**). However, toward the end of the twentieth century there was a growing recognition that large-scale planting efforts had had very limited success in terms of establishing viable mangrove forests (Primavera and Esteban, 2008). There was an emphasis within the scientific literature that mangrove restoration should be centered on restoring an area's ecological functions, such as re-establishing hydrological connectivity (Ellison, 2000; Lewis, 2005). Within our data, post-1990, planting based interventions continue at the same rate; however, case studies that incorporate particularly hydrologic and to a lesser extent hydraulic restoration started to be recorded (**Figure 5B**).

Of the 108 case studies where mangrove planting was the focus, there was almost an even split between those that used propagules (n = 23) compared to tree seedlings (n = 26). For 23 case studies the stage of planting material was not differentiated between propagules and seedlings and for the remaining 36 case studies that planted material was not reported. The most frequently planted species were those from the genera *Rhizophora*, *Avicennia* and *Bruguiera*, with *Rhizophora mucronata* Poir., *Avicennia marina* (Forssk.) Vierh. and *Rhizophora apiculata* Blume recorded in 17, 16, and 14 case studies, respectively. Ideally, the species planted should be driven by the location of the restoration site within the tidal frame and exposure and the aims of the project. For example, colonizing species such as *A. marina, A. alba* Blume and *Sonneratia alba* Sm. in the Indo West Pacific or *R. mangle* L. in the Atlantic East Pacific region may be used where restoration of fringing mangrove is aimed at providing storm protection (Primavera et al., 2011). While preferences on the species used were apparent and are supported by the published literature (see below), the variation between genera is also a function of their abundance. For instance, the families Caesalpinaceae, and Bignoniaceae, and the genus *Pelliciera* were not recorded in our case studies, which is likely a combination of their relatively restricted distribution in Central and South America (Spalding et al., 2010) and the few case studies identified from that region (**Figure 3**).

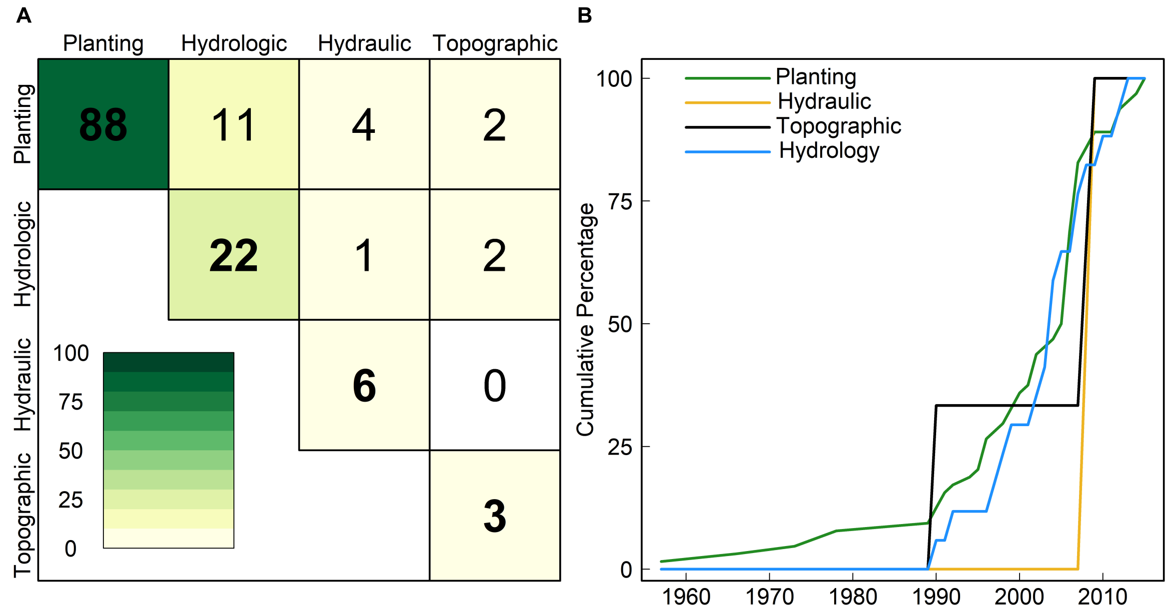

FIGURE 5 | (A) Frequency of different restoration methods used in the case studies. The diagonal boxes (bold numbers) identify the percentage of the different restorative methods used overall. The off-diagonal boxes represent the percentage of case studies that record information for the two interacting restorative methods. **(B)** The cumulative percentage of cases studies within each of the four intervention types vs. the start year of the intervention.

Mangrove planting efforts have been plagued by poor site/species matching, with species planted outside their physiological tolerances (Wodehouse and Rayment, 2019). Mangrove planting has generally been limited to the use of a restricted group of species, often from the genus *Rhizophora* (Friess et al., 2019). These species are preferred as they produce large propagules that are both easy to collect and are easily planted, and they exhibit fast growth (Samson and Rollon, 2008; Lee et al., 2019). *Rhizophora* species are also the most viable species for charcoal and firewood (Bandaranayake, 1998) so the establishment of a production forest may be seen as a co-benefit. It should be noted that the native species pool available to a restoration project varies hugely with location, with mangrove species richness highest (>25 native species) in the Indo-Malay Philippine Archipelago contrasting with communities limited to a single native species (Polidoro et al., 2010).

As well as relying on a small number of species, the majority of the mangrove replanting projects only use one or two mangrove species per site (Alongi, 2002; Wodehouse and Rayment, 2019). Out of the 108 case studies where planting occurred, 42 reported using a single species or species from a single genus, with a further 12 using two species or genera in the restoration. The remaining 48 case studies recorded between three and seven mangrove species or genera being planted. The use of monospecific planting has been questioned as it has been hypothesized that multispecies communities result in niche complementarity, and are needed to fully provide ecosystem services and functions (Kirui et al., 2008; Su et al., 2021). For example, it has been suggested that monospecific plantations of *Rhizophora* were more impacted following Typhoon Haiyan in the Philippines than other taxa (e.g., *Sonneratia*, *Avicennia*, and *Aegiceras*) as defoliated and damaged trees with broken stems were unable to regenerate new shoots resulting in high mortalities compared to the other taxa which had the potential to resprout epicormic buds (Villamayor et al., 2016). Monospecific restoration also often results in lower diversity of macrofauna and reduced habitat heterogeneity (Macintosh et al., 2002); however, outcomes of monospecific vs. mixed species restoration varies across species and ecosystem functions (Su et al., 2021).

Community Involvement

Out of the 123 case studies, 51 stated that local communities were involved during the restoration process. A large proportion of mangroves are found near rural communities (Aye et al., 2019) and mangroves support local livelihoods, through the provision of timber, fuelwood and food (Himes-Cornell et al., 2018). As such, community involvement in mangrove restoration is seen as a key determinant of restoration success—resulting in the development of concepts such as community-based mangrove management (CBMM). The CBMM approach has mainly been applied in countries where there was a wide scale adoption of decentralized governance policies, allowing local communities to take the initiative (Datta et al., 2012). Community led mangrove restoration projects generally have lower costs (Primavera and Esteban, 2008; Bayraktarov et al., 2016), with communities making decisions and conducting key tasks such as post intervention monitoring and governance (Brown et al., 2014; Mukherjee et al., 2015; Wylie et al., 2016).

Monitoring

Nearly two-thirds (n = 72) of the 123 case studies reported that some form of monitoring took place; however, only 19 case studies recorded the duration of that monitoring. The median length of post-restoration monitoring was 16 months, with the longest period recorded 9 years and 7 months, in line with the findings of Cadier et al. (2020).

The relatively short duration reported in the literature highlights a potential problem in that the time taken for an area to fully re-establish is often considerably longer than the monitoring timeframe. For example, in southwest Florida, 18 years after restoration, it was observed that the mangroves had similar measures to natural forests for certain factors such as species richness and vegetation cover, but were not yet comparable in terms of tree size and stem density (Proffitt and Devlin, 2005). Similarly, certain aspects of biodiversity and ecosystem function can take time to approach natural levels (Bosire et al., 2008). For example, 5–8 years after planting, sediment-infauna density and litter degradation in the mangroves of Gazi Bay, Kenya had not reached that of reference sites (Bosire et al., 2004, 2005).

The challenge of supporting longer-term monitoring was highlighted in a study from three states in India. Only 24% of the projects carried out monitoring of the restored area for 3 or more years (Mukherjee et al., 2015). Further, the majority of the projects that were carried out by foreign NGOs often did not include a monitoring section in the project design, resulting in the local communities banding together to monitor the area (Mukherjee et al., 2015). This issue is underpinned by the cost of long-term monitoring, with annual monitoring costs of 20% of the initial restoration budget observed in the Philippines (Primavera et al., 2012) and Vietnam (Tuan and Tinh, 2013). If accurate information on the location of the restoration site is available, there is the potential to remotely monitor mangrove extent change at regular intervals and lower cost using satellite imagery (Alexandris et al., 2013). However, there are challenges in detecting change in very small restoration areas without very high-resolution imagery.

Ecological Project Outcomes

Measures of vegetation structure are often recorded following restoration, as they are easy and rapid to quantify and there is assumption that vegetation recovers at a faster rate than an area's fauna or ecological functions (Ruiz-Jaen and Aide, 2005). In a review of indicators for coastal wetland restoration success, Cadier et al. (2020) identified metrics related to structural diversity as being the most frequently recorded, which for mangroves most commonly related to measures of mangrove density, height, diameter at breast height and basal area. However, even these simple structural metrics were infrequently recorded in our case studies. Less than 30% (n = 36) of the 123 case studies identified that natural recruitment had occurred at the restoration site, and despite the preponderance of mangrove planting (n = 108) within our case studies, only 47 recorded the percentage seedling survival. Future monitoring of coastal restoration should incorporate a greater use of metrics related to ecosystem functions (Bayraktarov et al., 2016; Cadier et al., 2020).

Socioeconomic Project Outcomes

To assess the socioeconomic outcomes recorded in a case study we used the typology of McKinnon et al. (2016). Socioeconomic outcomes were recorded from 31 case studies, with over half of these (n = 18) identifying more than one outcome (range: 1–5). The two most commonly cited outcomes were those related to material living standards and the economic living standards (**Figure 6**). In comparison, culture and spirituality, and freedom of choice and action were not recorded. Non-material values are hard to quantify and identify, and it is particularly difficult to quantify the link between changes in the non-material value of restoration interventions (Chan et al., 2012). Compared to other coastal ecosystems, mangrove restoration projects more frequently report socioeconomic outcomes (Bayraktarov et al., 2020). However, in the papers reviewed here that recorded socioeconomic outcomes, few attempted to quantify these. The value of mangroves to local communities has long been identified; however, quantifying the socioeconomic benefits of restoration is challenging. Understanding socioeconomic outcomes is key to improving the sustainable management of mangrove areas; for example, Satyanarayana et al. (2021) quantified the flows of money between charcoal and pole producers in the managed Matang Mangrove Forest Reserve.

Overall Trends

From our analysis several key trends were observed, which have the potential to impact our understanding of the magnitude and success of restoration within mangrove ecosystems. From our synthesis of the literature there was a clear spatial dominance in terms of the location with three quarters of the restoration taking place in Asia. This is perhaps unsurprising given the region's extensive mangrove forests (Spalding et al., 2010) and the multitude of human pressures impacting mangroves in this region (Richards and Friess, 2016). However, a significant finding was the general lack of case studies from large parts of Africa and Central and South America, limiting our ability to understand the drivers of restoration success and failure in those regions. The major causes of decline of mangroves in our restoration sites were linked to anthropogenic impacts, with losses due to agriculture and aquaculture particularly prevalent. The role of aquaculture in the decline of mangroves in Southeast Asia is well established (Richards and Friess, 2016), with these losses potentially explaining why so many restoration projects are occurring in this area.

Across our case studies, very few recorded a detailed breakdown of the project costs, and this coupled with limitations on the monitoring of outcomes from restoration, reduces our capacity to determine cost effectiveness of different interventions. This issue is not unique to mangrove restoration (Pienkowski et al., 2021), but without the greater implementation of standardized reporting frameworks (Spurgeon, 1999; Iacona et al., 2018) our ability to maximize the impact from limited conservation funding is diminished. One area where our synthesis suggests mangrove restoration has started to tackle more effectively is the integration of local communities within mangrove restoration. Over 40% of the case studies stated there was local community involvement to some extent. This progress may in part be driven by the more recent promotion of "community based ecological mangrove rehabilitation" (Brown et al., 2014) and the recognition

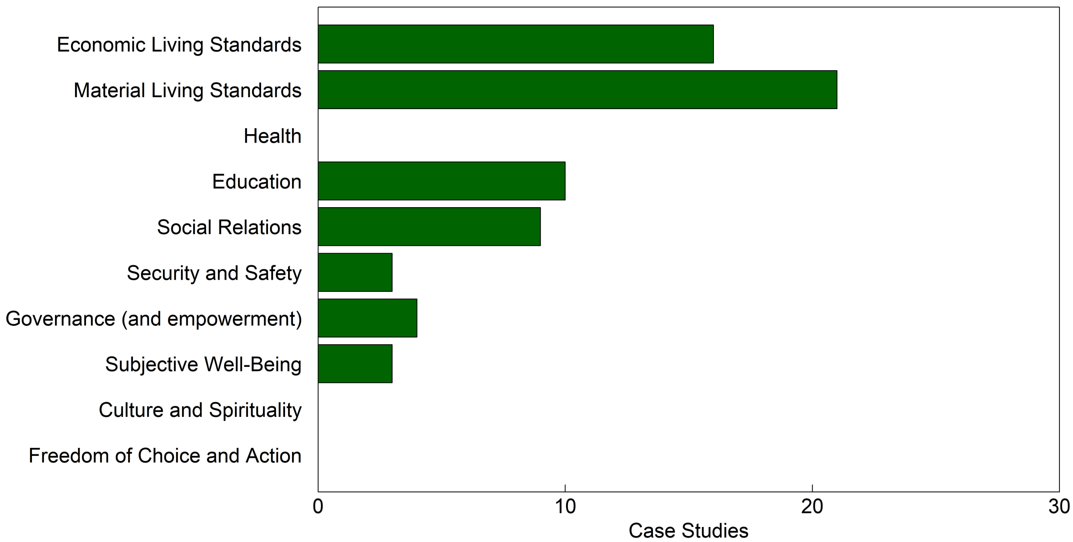

FIGURE 6 | The socioeconomic outcomes recorded in the restoration cases studies.

that mangrove restoration can have both ecological and socioeconomic outcomes (Bayraktarov et al., 2020). By adjusting mangrove restoration activities based on the needs and desires of local communities, and adequately supporting local community-led mangrove restoration it is likely that mangrove restoration will be more durable (Lovelock and Brown, 2019).

Our synthesis reaffirms challenges on several topics that have been repeatedly highlighted as limitations in mangrove restoration approaches. Firstly, the land tenure of a project location is often either unknown or not reported in nearly 90% of our case studies. This land tenure issue can often drive inappropriate restoration approaches that focus on planting mangroves in ecologically unsuitable locations (Lovelock and Brown, 2019). Secondly, for the majority of our case studies mangrove planting was still the intervention of choice. Large-scale, often monospecific planting of *Rhizophora* species, has been driven by unsuitable performance metrics and short-term measures of project success (Lee et al., 2019), and when coupled with land tenure issues and poor site/species matching this has resulted in limited mangrove survival and damage to other coastal habitats such as mudflats and seagrasses (Primavera and Esteban, 2008). Finally, the range of outcome metrics reported for restoration projects is large (Cadier et al., 2020), with the indicators recorded often not following a standardized methodology or without suitable comparison sites either in space or time. A more standardized reporting and monitoring framework would allow us to make inferences across restoration projects and different types of intervention, providing an understanding of the nuances of when and why restoration does or does not work, allowing organizations to adjust direct project implementation to improve outcomes and increasing our ability to determine cost-effectiveness. One area of promise is the integration of multiple types of outcome (e.g., economic, social, and ecological) with in the mangrove restoration literature, a trend not as apparent in other coastal, and marine ecosystems (Bayraktarov et al., 2020).

Overall, for certain sections of our framework, information in the published literature is reasonably well recorded (e.g., physical interventions and causes of decline), while several of the others lack all but the basic metrics. However, viewing all the sections through a single lens shows that there is lack of cohesion throughout. Unsurprisingly given journal styles and differences in paper focus, no paper follows the same reporting style and only reports a subset of the information. This results in information being of variable quality and consistency.

CONCLUSION

There has been a marked increase in marine restoration efforts over recent decades (Duarte et al., 2020), a trend which is likely to increase in response to multiple international, national and local policies associated with climate change mitigation and wider calls for "nature based solutions." For such restoration to have the maximum impact, both for biodiversity and for people, it is critical that we are able comprehensively and accurately to track restoration efforts. However, data collection in mangrove restoration projects has often been *ad hoc* and incomplete (Worthington and Spalding, 2018). Our analysis identified major knowledge gaps relating to the reporting of restoration costs and socioeconomic outcomes. We also found biases in location of restoration projects reported in the published literature, which have the potential to undermine restoration efforts in areas that are understudied.

Reporting key metrics and indicators on mangrove forest restoration, using standardized approaches, could provide a critical tool for restoration practitioners, both in evaluating their own efforts, and in providing a valuable baseline for future restoration. Given the increase in efforts there have been attempts to standardize the reporting and monitoring of restoration outcomes (e.g., FAO and WRI, 2019; Gann et al., 2019; Yando et al., 2021); however, challenges remain in implementing these global frameworks at the site level. There are also difficulties in applying generic frameworks to specific ecosystems given differences in project aims, restoration approaches, and potential outcomes. The review and synthesis of systematic reporting would also greatly facilitate current efforts to "scale up" mangrove restoration to the levels being targeted, including a likely reduction in failures, and considerable savings in terms of both reducing costs and optimizing benefits. A holistic approach capturing a broader set of measurements would facilitate an understanding of the environmental, socioeconomic and political setting, to inform what might be driving outcomes. Our findings show that there is a need for a framework which practitioners can use to report the process of their restoration studies and their outcomes. Such a framework would make it easier to compare across regions, approaches and outcomes allowing lessons to be learnt from previous restoration attempts.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author/s.

AUTHOR CONTRIBUTIONS

TAW, YMG, and MS conceived the idea. YMG undertook the literature review with input from TAW, DA-B, and PM. YMG wrote the manuscript with input from all authors.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/ffgc.2022.720394/full#supplementary-material

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