Research Topics

The mangroves of Southeast Asia in the United Nation's decade on ecosystem restoration

Edited by
Severino III Salmo, Kangkuso Analuddin, Richard MacKenzie and Sahadev Sharma

Published in
Frontiers in Marine Science

ISSN 1664-8714
ISBN 978-2-8325-4384-9
DOI 10.3389/978-2-8325-4384-9

About Frontiers

Frontiers is more than just an open access publisher of scholarly articles: it is a pioneering approach to the world of academia, radically improving the way scholarly research is managed. The grand vision of Frontiers is a world where all people have an equal opportunity to seek, share and generate knowledge. Frontiers provides immediate and permanent online open access to all its publications, but this alone is not enough to realize our grand goals.

Frontiers journal series

The Frontiers journal series is a multi-tier and interdisciplinary set of open-access, online journals, promising a paradigm shift from the current review, selection and dissemination processes in academic publishing. All Frontiers journals are driven by researchers for researchers; therefore, they constitute a service to the scholarly community. At the same time, the *Frontiers journal series* operates on a revolutionary invention, the tiered publishing system, initially addressing specific communities of scholars, and gradually climbing up to broader public understanding, thus serving the interests of the lay society, too.

Dedication to quality

Each Frontiers article is a landmark of the highest quality, thanks to genuinely collaborative interactions between authors and review editors, who include some of the world's best academicians. Research must be certified by peers before entering a stream of knowledge that may eventually reach the public - and shape society; therefore, Frontiers only applies the most rigorous and unbiased reviews. Frontiers revolutionizes research publishing by freely delivering the most outstanding research, evaluated with no bias from both the academic and social point of view. By applying the most advanced information technologies, Frontiers is catapulting scholarly publishing into a new generation.

What are Frontiers Research Topics?

Frontiers Research Topics are very popular trademarks of the *Frontiers journals series*: they are collections of at least ten articles, all centered on a particular subject. With their unique mix of varied contributions from Original Research to Review Articles, Frontiers Research Topics unify the most influential researchers, the latest key findings and historical advances in a hot research area.

Find out more on how to host your own Frontiers Research Topic or contribute to one as an author by contacting the Frontiers editorial office: frontiersin.org/about/contact

The mangroves of Southeast Asia in the United Nation’s decade on ecosystem restoration

Topic editors
Severino III Salmo — University of the Philippines Diliman, Philippines
Kangkuso Analuddin — Halu Oleo University, Indonesia
Richard MacKenzie — Institute of Pacific Islands Forestry, Pacific Southwest Research Station, Forest Service (USDA), United States
Sahadev Sharma — University of Malaya, Malaysia

Citation
Salmo, S. III., Analuddin, K., MacKenzie, R., Sharma, S., eds. (2024). *The mangroves of Southeast Asia in the United Nation’s decade on ecosystem restoration*. Lausanne: Frontiers Media SA. doi: 10.3389/978-2-8325-4384-9

Table of contents

TYPE Editorial
PUBLISHED 14 December 2023
DOI 10.3389/fmars.2023.1341796

OPEN ACCESS

EDITED AND REVIEWED BY
Laura Airoldi,
University of Padova Chioggia Hydrobiological Station, Italy

*CORRESPONDENCE
Severino G. Salmo III
sgsalmo@up.edu.ph

RECEIVED 20 November 2023
ACCEPTED 04 December 2023
PUBLISHED 14 December 2023

CITATION
Salmo SG III, MacKenzie RA and Analuddin K (2023) Editorial: The mangroves of Southeast Asia in the United Nation's decade on ecosystem restoration. *Front. Mar. Sci.* 10:1341796. doi: 10.3389/fmars.2023.1341796

Editorial: The mangroves of Southeast Asia in the United Nation's decade on ecosystem restoration

Severino G. Salmo III[1]*, Richard A. MacKenzie[2] and Kangkuso Analuddin[3]

[1]Institute of Biology, University of the Philippines Diliman, Quezon City, Philippines, [2]US Forest Service, Pacific Southwest Research Station, Institute of Pacific Islands Forestry, Hilo, HI, United States, [3]Halu Oleo University, Kendari, Indonesia

KEYWORDS

Southeast Asian mangroves, restoration, conservation, biodiversity, recovery of ecosystem services, climate change adaptation and mitigation

Editorial on the Research Topic
The mangroves of Southeast Asia in the United Nation's decade on ecosystem restoration

Introduction

Mangroves are recognized for several important ecological and socio-economic services they perform. The delivery of these services are directly linked to the livelihood and well-being of the societies that rely on mangrove forests, particularly on the provision of food and income and in protecting coastal human populations against the impacts of natural disasters (Sannigrahi et al., 2020). The largest and most diverse mangroves in the world are located in Southeast Asia (SE Asia), an area also considered to be a biodiversity hotspot (Bhowmik et al., 2022). Although global trends indicate mangrove gains in the last twenty years, mangrove losses are still reported in the region (Bryan-Brown et al., 2020). The causes of mangroves losses are not necessarily attributable to aquaculture ponds anymore (as was the case from 1970s to 1990s) but because of tremendous pressures for coastal reclamation/development, conversion to oil palm plantations, and to natural disasters (primarily typhoons and rising sea level (Hamilton and Friess, 2018). The region already has national-/internationally-awarded successful mangrove conservation and restoration programs as early as 1900s (Gerona-Daga & Salmo). However, the successes (or failures) of these programs are largely unreported and undocumented (Gatt et al., 2022; Lovelock et al., 2022). If only these previous programs have been properly documented, then the current and future restoration may have incorporated the lessons and avoided common causes of failures (Salmo, 2021).

The implementation of the United Nation's Decade on Ecosystem Restoration (2021-2030) provides an opportunity [but also a challenge (Waltham et al., 2020; da Rosa and Marques, 2022)] to reflect on previous lessons in order to advance mangrove restoration in SE Asia. For example, the 20 x 30 and 30 x 30 visions provide hopes to deliver conservation and restoration targets by year 2030. In this Research Topic, seven articles present the status and lessons, and provide perspectives for a "better" mangrove restoration strategies to help achieve the UN's targets/strategies on ecosystem restoration.

Restoration index and "bio-shields"

Juanico developed a "restoration index" to estimate the potential success of mangrove restoration programs in the Philippines. The index is a prospective tool that can assess the progress and success of restoration programs in terms of "bio-shielding" effect (especially in terms of coastal protection against catastrophic typhoons). The study further proposed that future restoration efforts should be moved further inland to have substantial forest. Restoring mangroves inland will be politically and socio-economically challenging as these are the same sites that are currently occupied by coastal residents and also targeted for future coastal development/reclamation programs. A substantial financial investment to restore the inland areas is needed.

Tracking undetected "historical mangrove losses" as indicator for selecting restoration areas

Baltezar et al. used a combination of several remote sensing approaches in tracking the "undetected" historical mangrove losses in Myanmar, Thailand, and Cambodia. The study showed specific pre-1990s mangrove maps gains and losses which the authors linked to the socio-political conditions and uncertainties in the three countries. This study not only established a new baseline that would better inform current understandings of mangrove change pre-1990s, but also helped in understanding the different needs that the people and their governments were trying to meet.

Valuing ecosystem services in conserved and restored mangroves

The systematic review of Lee et al on valuation of ecosystem services revealed the limited studies not just in Malaysia but is most likely the case for the entire region. Because of limited dissemination, the results of valuation studies lacks integration (and influence) in policies and governance. The authors recommended that future valuation studies in SE Asia should engage policy makers and incorporate a clear dissemination strategy (i.e., policy briefs on science-policy nexus).

Integrating "social capital", finance and policy in mangrove restoration

The lack or absence of restorable areas is one of the primary reasons why massive mangrove planting projects are conducted in sub-optimal areas (e.g., seagrass, mudflats, etc.). Shusheng et al. proposed "ecological bank" as an integrative restoration approach to attract more social investments and develop streamlined policies for mangrove restoration plans. The authors suggested that income that will be generated from industries using or located near mangroves will be used to support restoration projects and provide subsidies to pond owners and social investors.

Systematic assessment and monitoring of recovery of mangrove cover

Tinh et al. used a combination of satellite imagery analyses and field surveys in the assessment and monitoring of mangrove restoration projects over time. The authors clearly showed net gains and rates of increases (as hectare per year) at "commune level" in Mekong delta as well as in other provinces. Despite the perceived success, the authors raised concerns on: (1) narrow mangrove strips which could be vulnerable to rising sea level and coastal squeeze, and (2) conversion of mangroves to other land uses.

The eDNA as an adaptive biodiversity assessment tool

The stability and recovery of biodiversity are expected as one of the key outputs of mangrove restoration programs. It serves as evidence of recovery of ecosystem services especially when done in chronosequence and in comparison, with a reference and disturbed systems. But conventional biodiversity assessment and monitoring methods (e.g., plot/transect, field surveys, etc.) are very expensive and time-consuming. The eDNA technique has recently gained prominence in biodiversity assessment for most aquatic ecosystems but surprisingly is not widely adapted in mangrove yet. Wee et al. reviewed key technical and practical limitations but also provided several essential and practical guides to scientists, policymakers, conservation practitioners and mangrove forest managers in implementing eDNA metabarcoding as a biomonitoring tool in mangrove restoration programs.

Status, trends, and directions of mangrove restoration studies

Most of the mangrove restoration studies in the region were conducted in response to problems associated with conversion to aquaculture, coastal erosion, and natural disasters. Different countries have different foci based on national problems and priorities (Gerona-Daga and Salmo). A systematic assessment of impacts of restoration programs are rarely reported. Out of the available reports, the most commonly reported impacts are ecosystem functions that are directly related to the recovery of ecosystem services primarily "awareness" and "livelihood", but not the other equally important ecosystem functions. Research topics suggested in this study provide a path forward to improve mangrove restoration, and aid in the development of national and international restoration and conservation strategies. The authors further suggested that an international network among SE Asian scientists through the Association of Southeast Asian Nations (ASEAN) should be facilitated to come up with a more strategic mangrove research and management program.

Synthesis and recommendations: The Southeast Asian mangroves in 2030

The seven articles in this Research Topic provided an overview of different mangrove restoration approaches and programs in SE Asia (Figure 1). Most restoration programs were implemented to increase mangrove cover and as supplement to current conservation/protection programs. But considering that mangrove losses are still apparent and with threats from coastal reclamations and from natural disasters, mangrove restoration should no longer be considered as "supplement" rather should be a necessity that needs to be expedited (Beeston et al., 2023).

The United Nations' Decade on Ecosystem Restoration provides an excellent opportunity to highlight the importance of restoration in SE Asian mangroves. The estimated restorable areas [ca 334,000 ha sequestering ca. 8700 Mt CO_2e; cf (Worthington and Spalding, 2018)] including previously unaccounted damaged areas (Baltezar et al.) if successfully restored will put SE Asia as a model that will demonstrate the recoveries of ecosystem services. The challenge lies in providing empirical evidence that the restoration programs are successful based on "restoration indicators" (Gatt et al., 2022) including effectively restored areas, economic valuation, policy integration, etc. Hence the need for a more systematic and consistent monitoring and reporting following monitoring standards (Lovelock et al., 2022) to at least demonstrate that the biodiversity and restoration targets will be achieved in 2030.

Restoration programs will need to be moved more inland to ensure higher survival and reduce the threats of submergence from sea-level rise. Investments from each individual country will be needed to finance restoration projects but most SE Asian countries may not be able to afford. The global interest from companies and investors to finance mangrove conservation and restoration can be explored for funding support (Friess et al., 2022). The ASEAN can be tapped to facilitate the technical, policy and financial needs for the restoration programs in the region. Another opportunity is the presence of international institutions which have been providing technical and financial supports. The individual country's programs together with the facilitation of the ASEAN and with international institutions will need to be aligned to achieve the restoration targets in the region by 2030 (Figure 1).

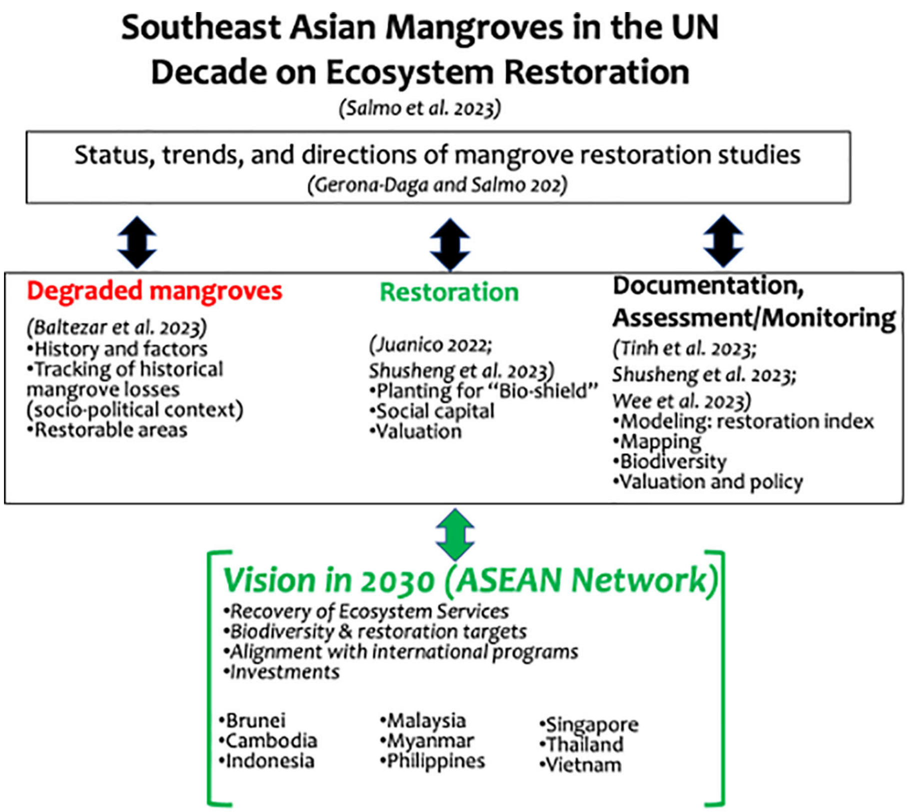


FIGURE 1
Summary of the seven manuscripts showing the status, trends, and recommendations to help achieve the mangrove restoration targets in 2030.

Author contributions

SS: Conceptualization, Formal analysis, Supervision, Writing – original draft, Writing – review & editing. RM: Writing – review & editing. KA: Writing – review & editing.

Acknowledgments

We thank the authors of the seven manuscripts who contributed in this Research Topic. SS and RM were supported by Partnerships for Enhanced Engagement in Research (PEER) under USAID cooperative agreement number AID-OAA-A-11-00012.

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.

Publisher's note

All claims expressed in this article are solely those of the authors and do not necessarily represent those of their affiliated organizations, or those of the publisher, the editors and the reviewers. Any product that may be evaluated in this article, or claim that may be made by its manufacturer, is not guaranteed or endorsed by the publisher.

References

Beeston, M., Cameron, C., Hagger, V., Howard, J., Lovelock, C., Sippo, J., et al (Eds.) (2023). *Best practice guidelines for mangrove restoration* (Washington DC: Global Mangrove Alliance). Available at: https://www.mangrovealliance.org/wp-content/uploads/2023/10/Best-Practice-for-Mangrove-Restoration-Guidelines-v2.pdf.

Bhowmik, A. K., Padmanaban, R., Cabral, P., and Romeiras, M. M. (2022). Global mangrove deforestation and its interacting social-ecological drivers: A systematic review and synthesis. *Sustainability* 14, 4433. doi: 10.3390/su14084433

Bryan-Brown, D. N., Connolly, R. M., Richards, D. R., Adame, F., Friess, D. A., and Brown, C. J. (2020). Global trends in mangrove forest fragmentation. *Sci. Rep.* 10, 7117. doi: 10.1038/s41598-020-63880-1

da Rosa, C. M., and Marques, M. C. M. (2022). How are biodiversity and carbon stock recovered during tropical forest restoration? Supporting the ecological paradigms and political context involved. *J. Nat. Conserv.* 65, 126115. doi: 10.1016/j.jnc.2021.126115

Friess, D. A., Howard, J., Huxham, M., Macreadie, P. I., and Ross, F. (2022). Capitalizing on the global financial interest in blue carbon. *PloS Clim.* 1 (8), e0000061. doi: 10.1371/journal.pclm.0000061

Gatt, Y. M., Andradi-Brown, D. A., Ahmadia, G. N., Martin, P. A., Sutherland, W. J., Spalding, M. D., et al. (2022). Quantifying the reporting, coverage, and consistency of key indicators in mangrove restoration projects. *Front. For. Glob. Change* 5. doi: 10.3389/ffgc.2022.720394

Hamilton, S. E., and Friess, D. A. (2018). Global carbon stocks and potential emissions due to mangrove deforestation from 2000 to 2012. *Nat. Clim. Change.* 8, 240–244. doi: 10.1038/s41558-018-0090-4

Lovelock, C. E., Barbier, E., and Duarte, C. M. (2022). Tackling the mangrove restoration challenge. *PloS Biol.* 20 (10), e3001836. doi: 10.1371/journal.pbio.3001836

Salmo, S. G. (2021). "Assessment of typhoon impacts and post-typhoon recovery in Philippine mangroves: lessons and challenges for adaptive management," in *Dynamic sedimentary environments of mangrove coasts* (Amsterdam, Netherlands: Elsevier), 539–562. doi: 10.1016/B978-0-12-816437-2.00022-7

Sannigrahi, S., Zhang, Q., Pilla, F., Joshi, P. K., Basu, B., Keesstra, S., et al. (2020). Responses of ecosystem services to natural and anthropogenic forcings: A spatial regression based assessment in the world's largest mangrove ecosystem. *Sci. Total. Environ.* 715, 137004. doi: 10.1016/j.scitotenv.2020.137004

Waltham, N. J., Elliott, M., Lee, S. Y., Lovelock, C., Duarte, C. M., Buelow, C., et al. (2020). UN decade on ecosystem restoration 2021–2030—What chance for success in restoring coastal ecosystems? *Front. Mar. Sci.* 7. doi: 10.3389/fmars.2020.00071

Worthington, T., and Spalding, M. (2018). *Mangrove restoration potential: A global map highlighting a critical opportunity* (Cambridge, England: Apollo – University of Cambridge Repository).

frontiers | Frontiers in Marine Science

TYPE Original Research
PUBLISHED 23 September 2022
DOI 10.3389/fmars.2022.968420

Check for updates

Does mangrove restoration imply coastal protection? A prospective simulation study

Drandreb Earl O. Juanico*

DataSc/ense TechnoCoRe, Technological Institute of the Philippines, Quezon City, Philippines

OPEN ACCESS

EDITED BY
Kangkuso Analuddin,
Halu Oleo University, Indonesia

REVIEWED BY
Ali Masria,
Mansoura University, Egypt
Ariel Blanco,
University of the Philippines Diliman, Philippines
Dixon Gevaña,
University of the Philippines Los Baños, Philippines

*CORRESPONDENCE
Drandreb Earl O. Juanico
reb.juanico@tip.edu.ph

SPECIALTY SECTION
This article was submitted to Marine Conservation and Sustainability, a section of the journal Frontiers in Marine Science

RECEIVED 14 June 2022
ACCEPTED 30 August 2022
PUBLISHED 23 September 2022

CITATION
Juanico DEO (2022) Does mangrove restoration imply coastal protection? A prospective simulation study. *Front. Mar. Sci.* 9:968420. doi: 10.3389/fmars.2022.968420

Mangrove restoration in the coastal zones is a concept proposed by environmental conservationists. Among the cited advantages of mangrove restoration are providing socio-economic services and coastal protection. Aware of these advantages, countries in Southeast Asia, such as the Philippines, have been implementing government- or civilian-backed restoration efforts. However, will current practices of restoration lead to the intended results? Also, are claims of coastal protection effects realistic? These two questions underscore the challenges posed by the long gap between the present intervention and future impact. Field evidence of protection may emerge from existing sites, the circumstances of which may not be easily portable onto other sites. This study examines the mangrove restoration practices in the Philippines and proposes the restoration index as a short-term prospective estimate of the future success of the restoration effort. This study also assesses the coastal protection potential of mangroves by examining the "bio-shielding" effect against storm surges driven by category-5 winds. Two coastal sites—Tacloban, Leyte, and Pan de Azucar, Iloilo—in the Philippines along the track of a category-5 storm, were considered. The restoration index was calculated based on the characteristics of *Rhizophora* mangroves commonly used in restoration programs. The coastal inundation model examined the extent of inland flooding due to storm surges by comparing an actual and hypothetical mangrove scenario for each site. A reasonable value of tree density obtained from the restoration simulations was estimated to determine if and to what degree, do mangroves in both sites offer coastal protection. For Tacloban, the actual mangroves are limited in scope, while the hypothetical scenario assumed a mangrove greenbelt fringing the city's eastern periphery. For Pan de Azucar, the existing mangroves are dense at the southwestern tip of the island, whereas in the hypothetical scenario, these mangroves are absent. The results, reinforced with a household survey, indicated a positive economic value of mangrove restoration for coastal protection. The restoration index and coastal inundation simulations are prospective tools that will guide the Philippines and Southeast Asia, in general, in formulating impactful mangrove restoration programs.

KEYWORDS

bio-shield, *Rhizophora*, mangrove restoration, coastal protection, storm surge

Introduction

Mangroves provide several socio-ecological and ecosystem goods and services such as timber and fisheries production, nutrient regulation, and shoreline protection [see a review by Lee et al. (2014)]. Unfortunately, mangroves are being lost worldwide at an alarming rate of 1% per year due to various natural and anthropogenic causes (FAO, 2007). In the Philippines, the total mangrove forest cover decreased by 51.8% between 1918 and 2010. Notably, an annual loss rate of 0.52% between 1990 and 2010 was mainly attributed to aquaculture development (Long et al., 2014). The depletion of mangroves may result in the reduction of ecosystem functionality and may increase the vulnerability of inhabited coastal plains to natural disasters such as storm surges (Duke et al., 2007).

In November 2013, Super Typhoon Haiyan ravaged the Eastern Visayas region in Central Philippines. At category 5 on the Saffir-Simpson hurricane scale, it is easily one if not the strongest in historical records that ever made landfall (Zhang, 2013; Holden and Marshall, 2018). In the quest for solutions to mitigate future coastal disasters, the protective capacity of mangroves along coastal fringes is being considered (Schmitt et al., 2013; Temmerman et al., 2013). Mangroves are known to attenuate waves such as storm surges by as much as 75% through its vast underground root networks and high structural complexity. However, this protective capacity is only viable if mangroves are dense across a vast area relative to the shoreline (McIvor et al., 2012). Coastal communities may thus benefit from mangrove restoration by enhanced protection against storm surge. Indeed, mangrove restoration provides many potential advantages to coastal communities, but assessing its long-term success, especially for coastal protection, has remained an open question.

The term “restoration” here is taken to denote an active, human-led effort to put a system back to a pre-existing condition claimed through historical evidence. But the efforts are done nevertheless whether or not the claimed past conditions were pristine (Lewis III, 2005). Stretches of Philippine coast were evidently populated by mangroves before urbanization accelerated, and before vast land conversions for profitable aquaculture ventures were made (Primavera and Esteban, 2008). However, quantitative evidence of mangrove restoration success is currently lacking because of weak support for the proposed rehabilitation programs, leading to only a few samples to consider. The dearth in support stems mainly from two sources of uncertainty. First, mangrove restoration has been characteristically open-ended, implying unpredictable outcomes (Kamali and Hashim, 2011). Second, success has not been measurable in the short term. Support, especially from the government, has required concrete assurance for returns on investment. Thus, a simulation study can be the only scientifically backed option to evaluate the prospective success of mangrove restoration. The state of ecological modeling of mangroves has improved quite substantially in the last two decades. Various models describing the biophysical characteristics of mangroves and their abiotic interactions have been proposed and validated. However, a focused study to assess restoration scenarios through a synthesis of mangrove simulation tools and link its results to coastal protection is yet to take root.

Real-time environment forecasting models have become possible because of enhancements in computational technology and improvements in numerical models. For example, the substantial increase in the accuracy of weather forecasting draws from the improvement in weather models supplemented by advanced satellite sensing equipment. With a mangrove-growth model (Salmo and Juanico, 2015), the current study assessed the mechanistic feasibility of mangroves for coastal protection, particularly against storm surge. The extent of damage by Haiyan in November 2013 heightened the scientific interest in coastal protection (Zhang, 2013).

Linking the efforts of mangrove restoration and coastal protection is uncommon in the literature to date. Either studies deal exclusively with evidence of the benefits of mangrove restoration (Temmerman et al., 2013; Su et al., 2021), or analyze directly the bio-shielding effect of mangroves using simulations (Zhang et al., 2012; Kamil et al., 2021) and field-based extrapolations (Delfino et al., 2015). A study that makes a more definite connection between the two has yet to be reported. The present study attempts to fill this gap with prospective simulations.

The study’s first objective is to assess whether existing restoration practices can achieve dense mangroves, which are expected to maximize the degree of coastal protection. The study’s second objective is to evaluate if the presence of reasonably dense coastal mangroves indeed provides bio-shield protection benefits. Addressing both objectives will clarify if mangrove restoration offers coastal protection, justifying public and private support for restoration efforts. The issue is especially relevant for the Philippines and Southeast Asia, where mangrove biodiversity is experiencing the most prominent loss rate and where vulnerability to extreme weather disturbances, such as storm surges, is significant.

Methodology

The prospective simulation study consists of two parts, namely, the mangrove restoration process and coastal bio-shield effect of mangroves. The first part is focused on projecting realistically the development of a restored *Rhizophora* plantation. The uncertainty of the model is accounted for by injecting stochastic population dynamics in the forward projections. The second part is focused on addressing whether or not a fully developed coastal

Rhizophora plantation (dominated by mature trees) can exert protection against the inundation damage of a storm surge driven by category-5 winds.

Numerical growth model

The mathematical growth model elaborated by Salmo and Juanico (2015) considered the following species-specific factors: α for the biological growth parameter, β for the biological leaf area index, the Malthusian growth rate Ω , and maximum diameter at breast height (DBH) D_{max} . This model is implemented numerically with Equation (1) by setting Δt equal to one day with the factors: $D_t \in \mathbb{R}$ for the plant's DBH at time t with $D_t \in [0.5,15)$ cm and $\Omega,\alpha,\beta,D_{max} \in \mathbb{R}$ with $\alpha \neq -2$. The functions $\sigma(x,y)$, $\eta(x,y)$, and $K_t(x,y)$ represent the stressor responses to salinity, inundation, and time-dependent competition, respectively. The input (x,y) is the location of the plant within an area of size $L \times L$. The types of landscapes and stressors associated with this area are described in the next section.

$$D(x,y,t+\Delta t) = D_t + \Omega\left(\frac{D_t^{\beta-\alpha-1}}{2+\alpha}\right)\left[1-\left(\frac{D_t}{D_{max}}\right)^{1+\alpha}\right]\sigma(x,y)\ \ \eta(x,y)\ \ K_t(x,y) \tag{1}$$

The equation is based on the Malthusian (exponential) growth model and modulated by the stressor responses. The landscape for the σ, η, and the K competition field was changed.

Geometry

Figure 1A illustrates the landscape being modeled with slope parameter sl=0.52 , while Figure 1B displays the corresponding inundation stress field of this landscape patch. The landscape was a $L \times L$ patch with the diagonal representing the coastline. By default, this diagonal is the mean sea level (MSL). Both the functions $\sigma(x,y)$ and $\eta(x,y)$ have the input parameter sl for the slope of the terrain. The slope describes the reach of the mean high tide (MHT). For this setup, the MHT can reach approximately halfway inland as shown by the orange dashes in Figure 1A. Since none of the trees present are within reach of the MHT, the corresponding inundation stress at their location is effectively zero, as indicated in Figure 1B. Consider a landscape with sl=0 as illustrated in Figure 1C where the MHT reaches the left corner of the patch at (0,0) , while Figure 1D shows the inundation stress field as an increasing linear function starting with a value of 0 from the origin (0,0) to 0.5 at the MSL line (diagonal). For this case, the whole patch is submerged during high tide.

Figures 2A, B elaborate further what sl=0 means in terms of inclination. Let L=40.0 meters, then side b=28.3 meters. If the difference between the MSL and MHT is 1 meter, then the inclination of the landscape, which is the angle between c and b in Figure 2B is about 2.05 degrees. Hence, the landscape modeled by sl=0 has the terrain underwater by a depth of 1 meter at the coordinate (40,40) . Consider the case represented by Figures 2C, D. If d=14.1 such that the MHT reaches halfway inland, then the inclination is 4.05 degrees. A case where the

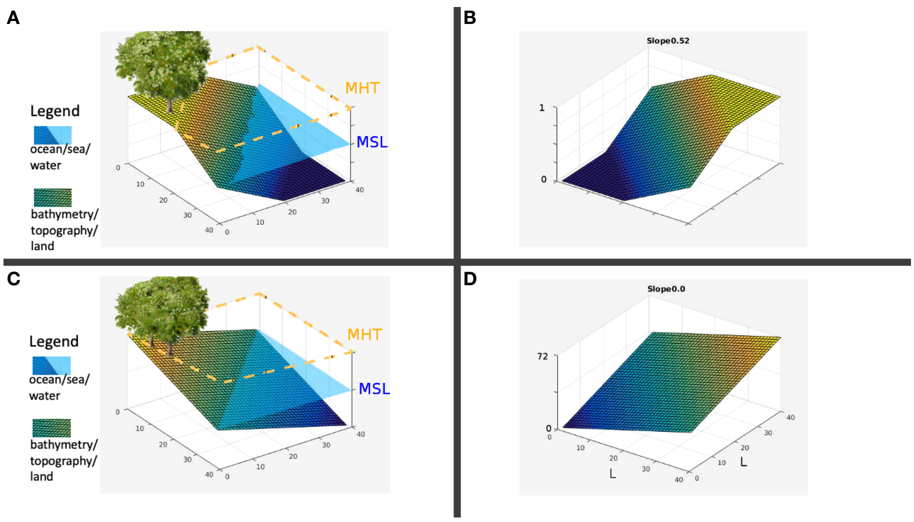


FIGURE 1
(A) Sample actual landscape; **(B)** Spatial value for inundation; **(C)** Slope, sl=0.01 landscape; **(D)** Inundation values.

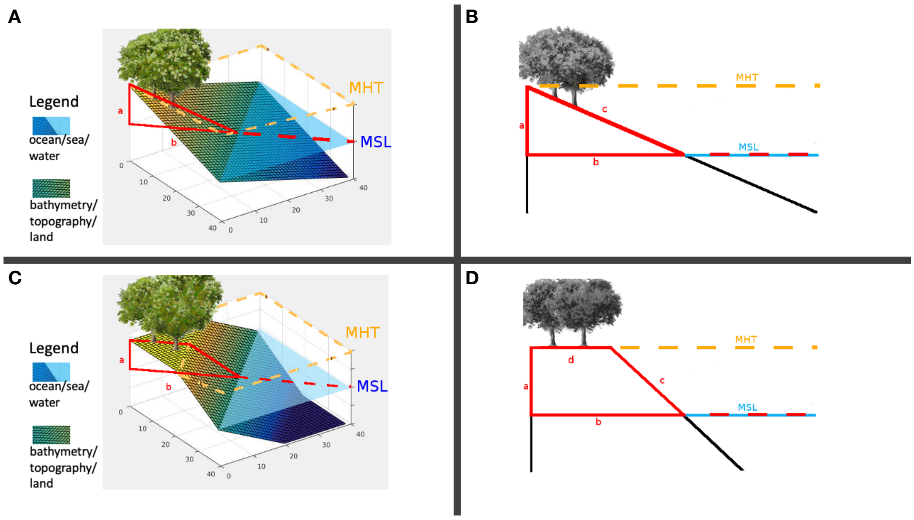


FIGURE 2
(A) Landscape patch, *sl*=0 ; **(B)** Inclination, *sl*=0 ; **(C)** Landscape patch, *sl*=0.5 ; **(D)** Inclination, *sl*=0.5 .

MHT fails to penetrate inland is instantiated by Figure 3, resembling a coastal cliff with slope value *sl*≈1 .

Stressors

The models of the salinity field *S*(*x*,*y*) and inundation field *I* (*x*,*y*) are embedded on the landscape. Stressor fields (Supplementary Material) are treated as piecewise functions of the slope parameter *sl* . The salinity field consists of values between 0 and 72. This field then influences the salinity response according to the following:

$$\sigma(x,y) = \{1 + \exp[ds(U_i - S(x,y))]\}^{-1} \tag{2}$$

For a given slope, a plant located at (*x*,*y*) will encounter inundation stress according to another piecewise model. The inundation response is determined by the following:

$$\eta(x,y) = 1 - I(x,y) \tag{3}$$

For the individual plant competition (Supplementary Material), the model used was the FON approach, which sums up the single field intensities of neighboring trees into an aggregate field strength *F*(*x*,*y*) (Berger and Hildenbrandt, 2000). In FON, it is assumed that individual growth is impossible if the quantity referred as "strength of

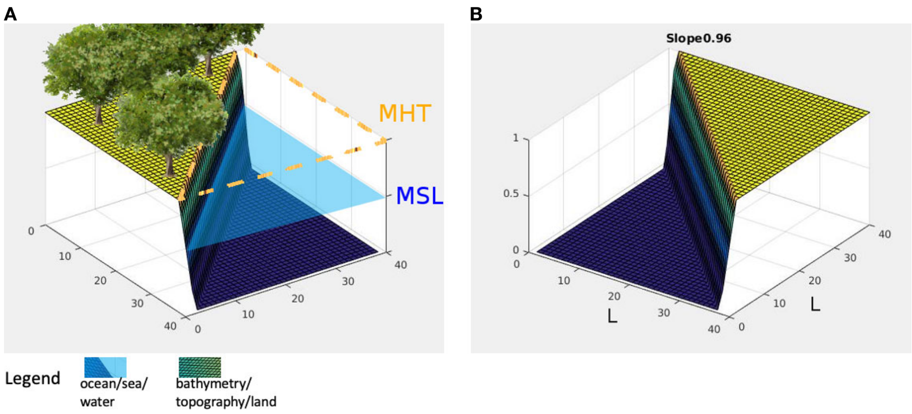


FIGURE 3
(A) Landscape patch, *sl*=1 ; **(B)** Inundation values for *sl*=1 .

neighborhood" F_A exceeds 0.5. Therefore, the growth of the plant is stopped when F_A>0.5 , i.e., when the function $C(F_A)$=1 $-2F_A$ is less than zero.

Stochastic population dynamics

The change in the number of individual plants at different life stages is described by a stochastic compartmental model (Supplementary Material). The demographic events consist of recruitment, mortality (Schaal and Leverich, 1982) and growth (Fulton, 1993), which may be expressed as state-transition equations.

- Recruitment: Tree $\xrightarrow{r}$ Tree + Seedling
- Seedling death: Seedling $\xrightarrow{m_{\text{seed}}}$ Dead
- Sapling death: Sapling $\xrightarrow{m_{\text{sap}}}$ Dead
- Tree death: Tree $\xrightarrow{m_{\text{tree}}}$ Dead
- Seedling to sapling growth: Seedling $\xrightarrow{g_{\text{seed}}}$ Sapling
- Sapling to tree growth: Sapling $\xrightarrow{g_{\text{sap}}}$ Tree

The growth rates g_{seed} and g_{sap} of seedling and sapling, respectively, can be estimated by the DBH growth rate across the defined size at the transition: 2.5 cm and 5.0 cm. In this manner, the practical measurement of the growth rates can be made from field surveys.

The state-transition equations are implemented with a stochastic simulation algorithm formulated by Gillespie (1976). Recruitment was established as seeds take root outside the crown of any existing tree. The dispersal rate multiplied by the time for the next event gives the maximum distance a seed can travel from the parent tree (given that seeds float in seawater).

Restoration index

With linear stability analysis (Supplementary Material), the average, long-term dynamical behavior of the stochastic model can be described with the expansion method by Van Kampen (1992). From this average (sometimes referred as "mean-field") dynamics, a system of ordinary differential equations can be examined further for its bifurcation properties. A dimensionless parameter ξ , reminiscent of the basic reproduction number in epidemic models, can be derived from the transition rates of the stochastic model. The result is the following index:

$$\xi = \frac{r \quad g_{\text{sap}} \quad g_{\text{seed}}}{m_{\text{tree}} \quad m_{\text{sap}} \quad m_{\text{seed}}} \tag{4}$$

By determining the value of the transition rates over one or two years from a trial plantation in a particular site, then ξ could be estimated. The restoration index (Equation 4) has a critical value equal to one, which is the value that separates the average, long-term behavior of the model into two: (1) endemic equilibrium if ξ>1; and (2) extinction equilibrium if ξ<1. The endemic equilibrium is the state described by the survival of the plant population long after it was established. The extinction equilibrium is the opposite state in which the population died out eventually. The endemic equilibrium is the preferred outcome of restoration.

Sensitivity analysis

The sensitivity analysis aims to get a good measure of how a time series diverges given an elementary effect. Particularly, the sensitivity of the following outcomes were analyzed:

- Population after 25 years
- Maximum population achieved
- Time when the maximum population was achieved
- Time of first tree appearance

The behavior of the stochastic model can be affected by the numerical value of the factors given in Equation 1 (namely, α, β, Ω) and the slope sl , which are implicit in Equation 2 and 3. Although considered as constants, U_i and ds in Equation 3 are likewise included in the sensitivity analysis. The ranges of the parameter values are:

- $\alpha \in [0.80,1.20]$ for the species-specific growth parameter
- $\beta \in [1.50,2.50]$ for the leaf area index
- $\Omega \in [0.01,0.25]$ for the Malthusian growth rate
- $Sl \in [0.1]$ for the slope parameter (hereinafter, designated as "slope")
- $U_i \in [0,100]$ for the trigger of the salinity stressor
- ds [-0.75,0.25] for the effects on the salinity gradient (hereinafter, designated as "dsalt")

The above sample ranges were restricted to values within reasonable exploration. For example, Ω>0.25 would model mangroves capable of maturing to tree status in less than three months. Using the framework of elementary effects analysis (Supplementary Material), the partial difference with respect to a single factor of the model must be averaged (Morris, 1991). This averaging yields an effect with mean μ (taken as an absolute value) and standard deviation σ . A high μ suggests that the concerned factor generally shifts the outcome of the model by a large degree. A low σ implies that the concerned factor indeed affects the outcome of the model.

Bio-shield simulations

The effect of coastal mangroves to the inland propagation of storm surge was simulated in quasi-3D, which builds on earlier quasi-2D simulation results for wave-vegetation dynamics (Zhang et al., 2012). The simulations involved a combination of several models of the relevant physics and biology of the coastal vegetation system (Figure 4). The physical part accounted for the water flow and the spatial profile of the sea floor (bathymetry) and land (topography). The biological aspect accounted for the fully developed mangrove along the coastline, representing its interaction with incoming sea waves by the tree density, average height, and average DBH (Suzuki et al., 2012). This parametrization effectively interfaced the result of the mangrove restoration simulation with the coastal wave model using *XBeach* version 1.22.4714:4905M (Roelvink et al., 2015). The model considered a simplified three-section vertical structure for the individual *Rhizophora* tree with a crown, trunk, and prop roots (Supplementary Figure 1). The tree density (per ha) from the restoration simulations were taken as vegetation input for the coastal wave model. For Tacloban, Leyte, due to the thin existing mangroves, the assumed tree density was 125/ha. For another site in Iloilo, the assumed tree density was 1,250/ha based on field observations. Both assumed tree densities were lower than the values projected from the restoration simulations. This underestimate was intended to buffer for any overestimation arising from the restoration model.

To describe the storm surge development, propagation and impact, three factors of the wave dynamics of Haiyan (Supplementary Material) were considered: wave conditions, including wind direction; bathymetry and topography, including the coastline orientation; and demographic landscape. The conditions of Haiyan were simulated from the available wind data. The bathymetry and topography were reconstructed from available NASA SRTM dataset (jpl.nasa.gov). The model bathymetry was generated using the following steps with *Delft3D*. First, a grid was defined using spherical coordinates spanning the area of interest. Second, with RGFGRID the grid was orthogonalized and refined to obtain a median pixel size of about 40m×40m . Then, with QUICKIN, a triangular interpolation was used to generate the model bathymetry from SRTM15+ data with an intrinsic resolution of 500m×500m at the equator (Tozer et al., 2019). The SRTM15+ dataset has also been utilized for inundation models with reasonably satisfactory results in tsunami simulations (Serra et al., 2021; Qiu et al., 2022) and sea-level rise (Wang and Marsooli, 2021). Lastly, the population census statistics during the period of the Haiyan disaster in Tacloban, Leyte were used to estimate the spatial density of human settlements in the urban area.

Field mangrove observations

Due to the dearth of mangroves in Tacloban, Leyte (Carlos et al., 2015), a different location with a dense coastal mangrove and along Haiyan's path was sought. A site in the northeastern Iloilo, Pan de Azucar Island (also known locally as Tambaliza) that belonged to a town called Concepcion, fits the description (Figure 5). The mangrove of the island was mapped and encoded in the bio-shield simulations. Particularly, the estimate of the tree density, estimated at around 1,250/ha, was a crucial input to the simulations. The simulation results for Pan de Azucar was used as a baseline to verify if endemic mangroves exerted any appreciable coastal protection effect against the category-5 disturbance brought by Haiyan.

Household surveys

An effective bio-shield must offer protection to a populated town from storm surge damage, while minimizing the cost of establishing this bio-shield. Tacloban was the town in the Philippines that suffered the most damage by Haiyan in 2013.

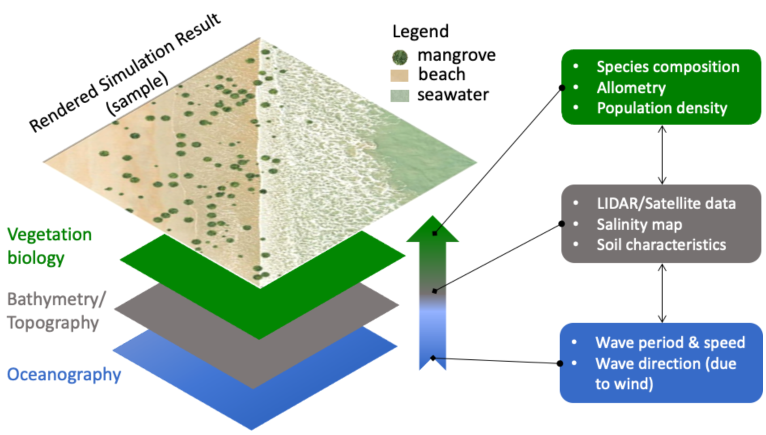


FIGURE 4
Framework of the bio-shield simulation. Bottom to top are the data layers essential for building the model. Oceanography and bathymetry/ topography make up the physical part, while vegetation biology makes up the biological part of the model. Each layer offers to the model the relevant fundamental variables (as shown on the right). The upward pointing arrow represents the bottom-top combination of these variables in the simulations, rendering a result such as the one shown at the topmost frame.

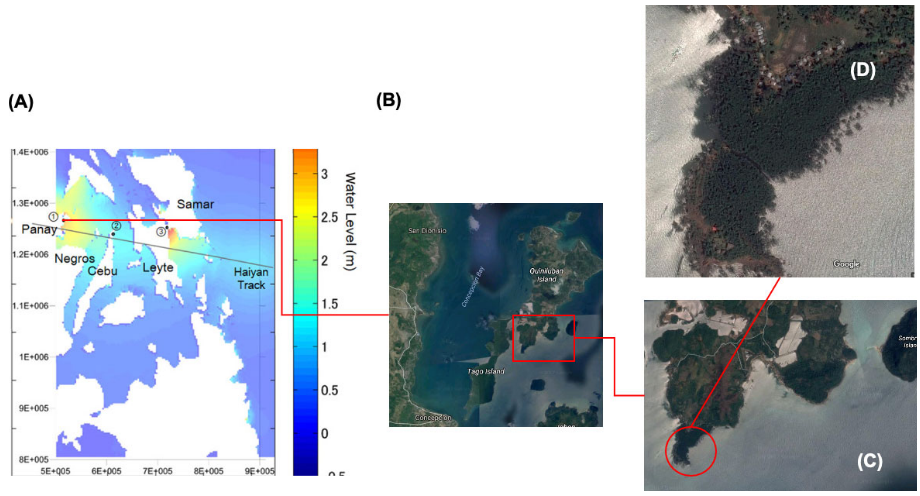


FIGURE 5
The site of Concepcion, Iloilo within the path of Haiyan on November 8, 2013. **(A)** Map showing part of the Philippines along the storm track and the relative location of Concepcion. **(B)** Satellite image of the Pan de Azucar island with the area of interest marked by the rectangle. **(C)** Larger view of the area of interest with the dense mangrove marked by the circle. **(D)** Larger view of the dense mangrove.

The economic viability of establishing the bio-shield needed an estimate of the value at risk for Tacloban. A household survey was deployed across the city between December 2016 and February 2017, which gathered data from a convenience sample of 5,000 households scattered across Tacloban's metropolitan area of which 55.7% responded (Supplementary Table). The survey sought to estimate how much economic value was at risk of inundation damage by the Haiyan storm surge with the following index:

$$HV = \text{GDP per capita} \times \text{Number of household members} + \text{Tangible Assets} \quad (5)$$

Equation 5 tags a value of every member as equivalent to the GDP per capita in Tacloban. This value represented the average income or potential income that the household member would contribute to Tacloban's economy. The tangible assets included the detachable possessions such as household appliances and other valuable items. HV, thus, estimated the casualty value that Tacloban would lose due to a sudden catastrophic disturbance. The same estimate was applied to Pan de Azucar.

Delimitations

The HV index in Equation 5 did not attempt to place a value on the real property for data privacy reasons. The economic valuation of the bio-shield also did not consider the carbon storage benefits of mangroves. The spatial resolution of the SRTM15+ bathymetry is not the highest available. However, due to computational limitations, this dataset was chosen. For further simplicity, the bio-shield simulations did not include sea level rise due to the melting of the polar ice caps. Studies have shown that a reasonable rate of sea level rise in Philippine coasts is about 15 mm/y, which is about nine times the global average (Holden and Marshall, 2018). At this rate, MSL would recede by about 38 cm in 25 years, which the present study considers small enough to assume a quasi-steady coastline over the course of the simulations. Lastly, the mangrove restoration and bio-shield simulations only assumed *Rhizophora* mangroves to simplify the estimates of growth rates and drag coefficients. The simulated patch is large enough (L=40 m) to generate an extended mangrove, but sufficiently small to manage the computational memory requirements.

Results

Sensitivity analysis

Based on multiple runs of the stochastic simulations, the Malthusian parameter Ω, and salinity response parameters dsalt and U_i exerted the most influence on the long-term tree population sizes (Figures 6A, B). Figure 6C also shows that Ω exerted substantial influence on the plant's time to maturity, which is essential for establishment and survivability. A separate analysis for the time derivative of Equation 1 was made for the factors Ω, α, β, and $D\in[0.5,15]$ cm. The increments for Ω, α, and β were maintained including their respective ranges with the exception of D, where it was scaled by a factor of 1.45. The domain of the random samples from the functions σ, η, and K was $[0,1]$. With the factors dsalt and U_i omitted, the curvature of

σ did not play a role in analyzing the elementary effects of the factors in consideration. Figure 6D shows the result of a million samples taken from the factor space, with Ω also exerting the strongest influence among the factors even though the scale was 10^{-3} . The slope relatively exerted a moderate effect on the long-term outcomes, while the model was least sensitive to α and β suggesting its robustness to species-specific variations (e.g., among different *Rhizophora* species) and foliage cover.

Mangrove restoration

Due to stochastic dynamics, the result of each simulation is a time series of tree, sapling, and seedling population taken as the average of ten runs. A typical outcome of a restored mangrove after a run time of 25 years consists of a distributed population across a flat coastal landscape (Figure 7). The initial seedling plantation (Figure 7A) generates the distribution of mangroves 25 years post-planting (Figure 7B). The landscape-rendered distribution is shown in Figure 7C. The youngest plants are mostly situated at the frontline along the MSL, while a few can be found in spaces between mature trees.

The time evolution of the population size based on tree, sapling, and seedling compartments are shown in Figure 8 for the two extreme slope cases, sl=0 (tidal flat) and sl=1 (coastal cliff). Stressor gradient effects are apparent on for sl=0 , in which the maturity is delayed on some batches because of the seaward increase of stress due to inundation and salinity. For the case sl=1 , the majority of initial seedlings reach their sapling and tree stages at the same time in the absence of salinity and inundation stressor gradients. The seedling population dropped quickly but not to zero, as seedlings planted in sites where stressors are high go through stunted growth. This gap is consistent with the gap between the number of seedlings before the drop and sapling population immediately after. The spike in the tree population near year 5 corresponds to a batch situated near the edge of the cliff. The peak tree density (considering the area approximately 0.08 ha) are 9,500/ha and 11,900/ha on the tidal flat and coastal cliff, respectively. At 25 years post-planting, the tree densities are about the same, at around 3,500/ha. The consistent decline in tree density after an initial peak indicates population thinning due to competition. It involves the DBH growth facilitated by the opening of space due to the death of some trees.

An extended simulation of a particular restoration effort (Figure 9A) reveals that the average tree density (Figure 9B) tends to increase in the long term after it peaks then declines in the medium term (Figure 9C). This average result appears to be guaranteed if ξ>1 in the first few years post-planting (Figure 9D). Furthermore, moving the restoration zone inland can improve long-term outcomes as can be illustrated with a time series (Supplementary Figure 2) or a phase portrait (Supplementary Figure 3). The restoration index, like the basic reproduction number in epidemic models, provides a short-term gauge of the effort's long-term success in obtaining dense mangroves. Considering coastal protection, it is necessary that the coastal mangroves have a high tree density over the long term to serve as bio-shield against storm surges with high, but less common, intensity.

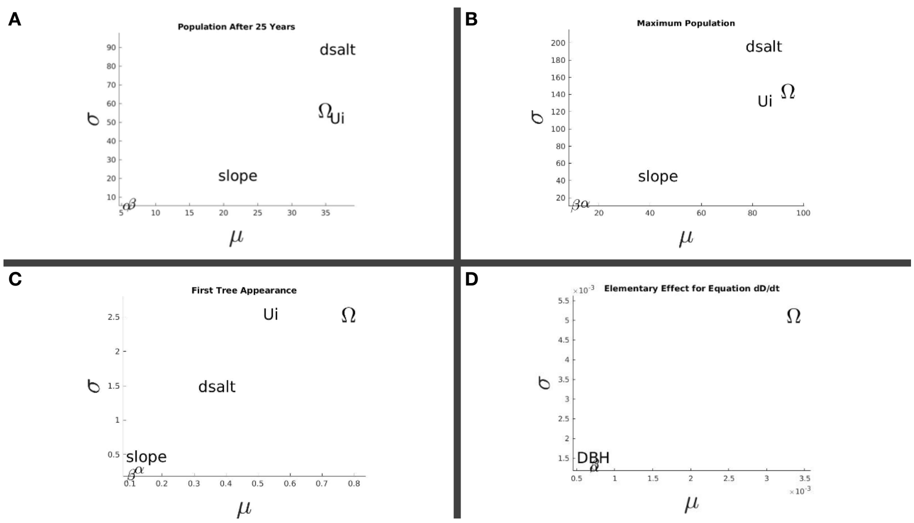


FIGURE 6
Sensitivity analysis on the long-term restoration outcomes. Sensitivity to selected parameters (see Section 2.6) of: **(A)** the average tree population 25 years post-planting; **(B)** the highest tree population achieved; **(C)** the time post-planting when the first instance of a plant maturing as a tree occurred; and **(D)** the elementary effect on the rate of DBH increase.

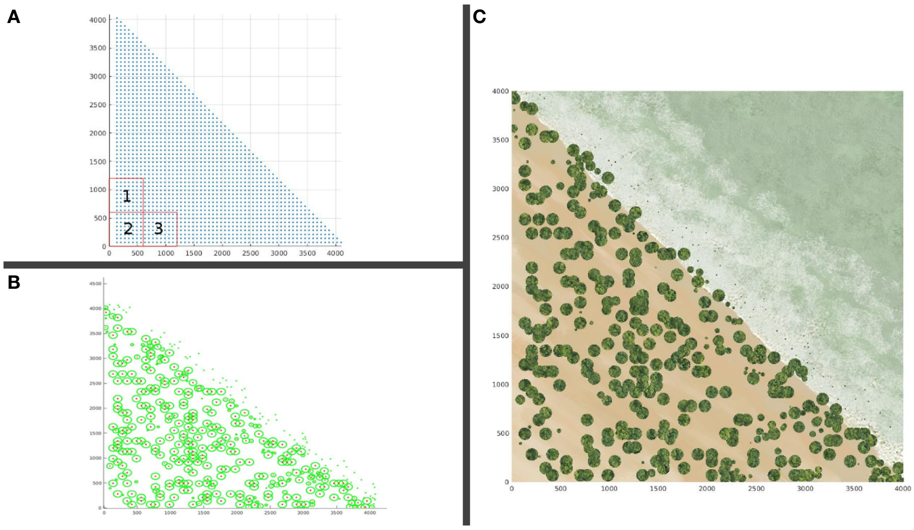


FIGURE 7
Typical result showing crowns of individual mangroves as viewed from the top for a landscape patch with slope parameter, sl = 0. **(A)** Patch plot at time zero; **(B)** 25 years post-planting, showing crown and trunk overhead; **(C)** Rendered result showing the beach landscape.

Bio-shield simulations

The coastal inundation results for Tacloban corroborated with field measurements conducted about two weeks after the storm (Lagmay et al., 2015). The water level results from the simulations of the actual scenario for Tacloban in the San Jose peninsula and downtown area (Figure 10) agreed with field data. The water level due to the surge was between 4-5 m in San Jose, whereas it ranged from 5 to 6 m in the downtown area (Lagmay et al., 2015). The simulation results at the peak of the surge (Figure 11A) indicated water levels within similar ranges in San Jose and downtown. This corroboration was sufficient to assume that the model represented coastal inundation to an appreciable degree.

The primary purpose of the study was to determine if mangroves could have protected Tacloban, Leyte when Haiyan generated intense storm surges. For this purpose, it is important to consider that the dissipative effect of vegetation assumed in *XBeach* has been validated from extensive laboratory and field tests (Roelvink et al., 2015). Tacloban, by default, did not have mangroves that Pan de Azucar had during Haiyan. The test scenario for Tacloban, thus, considered hypothetical mangroves. The second question was in which parts of the coast would restoration efforts be situated. Given the complexity of interaction between the wind profile and Tacloban's bathymetry and topography, this question was not trivial to address. The solution was to provide an optimal and economically viable reason for coastal mangrove restoration.

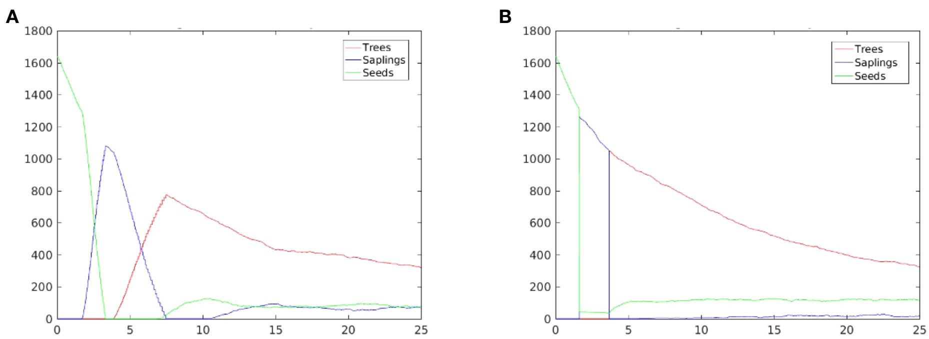


FIGURE 8
Prospective mangrove population curves resulting from beaches with different slopes. **(A)** slope parameter, *sl*=0 ; and **(B)** *sl*=1 .

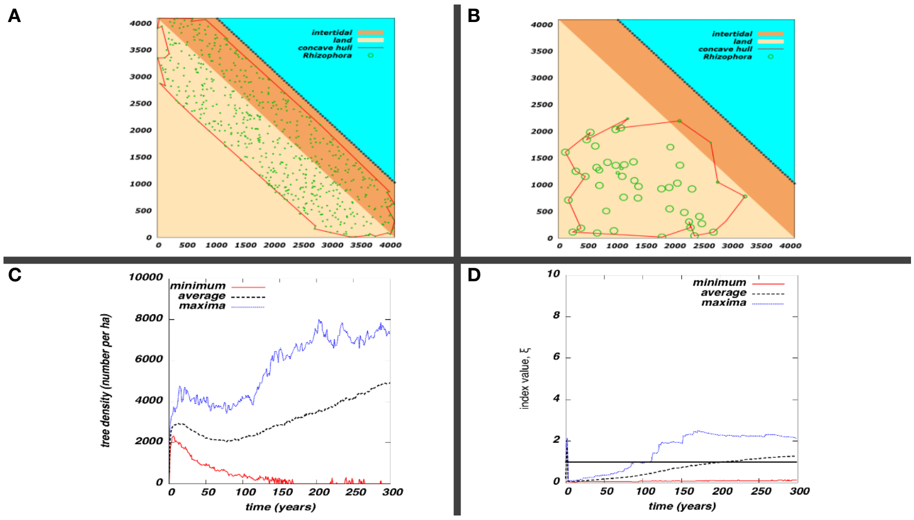


FIGURE 9
Restoration effort performance. **(A)** Initial seedlings distributed across a designated area (red polygon as convex hull); **(B)** the spatial distribution of plants 300 years post-planting; **(C)** Maximum, average, and minimum tree density versus time post-planting; and **(D)** Maximum, average, and minimum of the restoration index versus time post-planting. For **(C, D)**, the data were obtained from 100 stochastic realizations of the model.

The restoration strategy was to minimize the total area occupied by the mangroves, which should minimize the relocation of existing households. The intended outcome was to maximize the area that the storm surge could not inundate, which translated to maximizing the coastal protection capacity.

The hypothetical mangroves, which directly face Haiyan's wind direction, are located around the San Jose Peninsula toward the southeastern coast (Figure 10). A thin mangrove exists on the northern tip of the San Jose Peninsula. Simulating the Haiyan storm surges over Tacloban in November 2013 (Figure 11A), the hypothetical mangroves do not completely shield Tacloban totally from the inundation (Figure 11B). The incoming storm surge could still wash over San Jose peninsula, although the mangroves slowed down the transmitted water flow moving toward the southwestern portion of Tacloban (Supplementary Movie 1). Consequently, areas in the southeastern sector of Tacloban are notably drier. This southeastern sector coincides with the location of the San Jose,

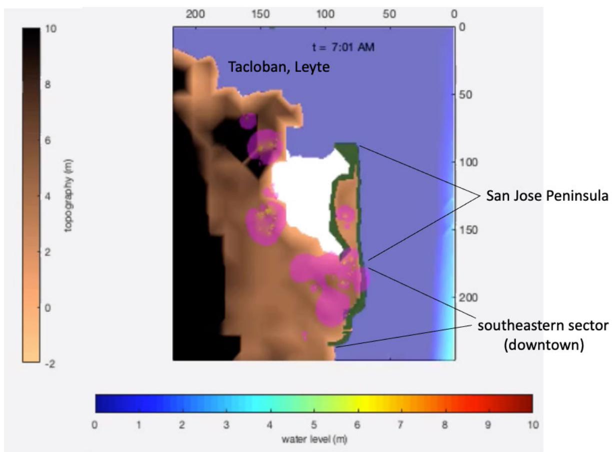


FIGURE 10
Site of mangrove restoration in Tacloban. The pink areas are density estimates of the concentration of households based on a household survey. The area occupied by households is widest in the southeastern sector of Tacloban, which is consistent with the latest population census from the Philippine Statistics Authority.

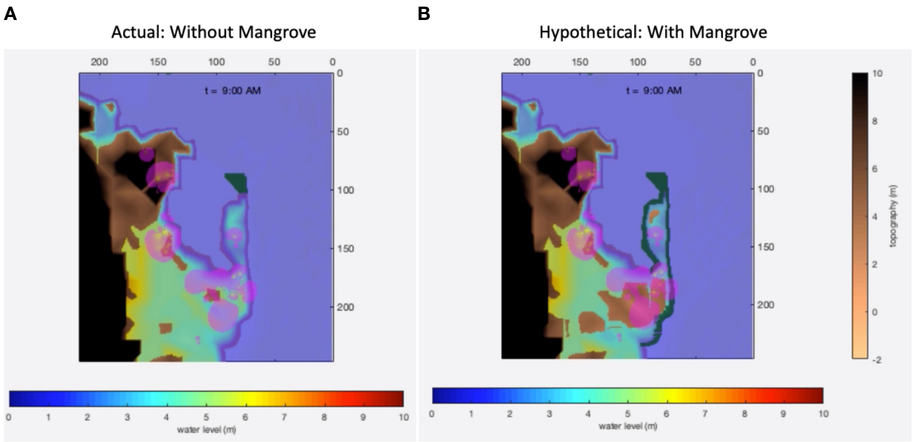


FIGURE 11
Bio-shield simulations in Pan de Azucar Island of the town of Concepcion, Iloilo. **(A)** Existing conditions with mangroves; **(B)** Hypothetical conditions without mangroves. The blobs (pink) represent the household population density estimated from the location data collected during the household survey.

Caibaan, and Marasbaras villages, which is home to about 5,500 households.

The presence of mangroves in the Pan de Azucar Island of Concepcion, Iloilo enabled testing the protective bio-shield effect (Figure 12A) in comparison to a hypothetical absence of mangroves (Figure 12B). This comparison is the exact opposite to the one applied to Tacloban—a converse hypothesis test. The existing mangroves were facing an oblique direction to Haiyan's wind velocity. Simulating Haiyan conditions revealed that the inland inundation would have been wider and deeper in the absence of mangroves, possibly penetrating into Sitio Proper (See and Wilmsen, 2022). The 1,250/ha mangroves in the surveyed area lessened the momentum of the reflected waves that penetrated inland, acting like shock absorbers that limited the energy of water flow (Supplementary Movie 2). This physical effect may also be a consequence of the oblique "angle of attack" of the winds relative to the mangrove orientation, although this aspect may require further confirmation. However, this result would suggest that the bio-shielding impact of mangroves may be site specific. Thus, the protection capacity of mangroves, which ground-truth reports in the area corroborated, were a factor in the relatively lower degree of damage and per-capita casualty rates in Pan de Azucar relative to Tacloban during the Haiyan storm surges in November 2013.

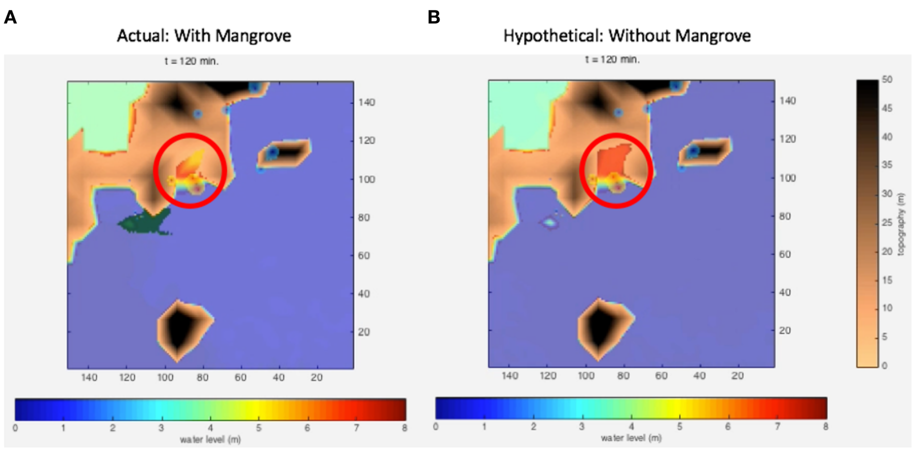


FIGURE 12
Bio-shield simulations in Pan de Azucar Island of the town of Concepcion, Iloilo. **(A)** Existing conditions with mangroves; **(B)** Hypothetical conditions without mangroves. The encircled inland area shows the effect of the absence of mangroves that currently exist to the southwest.

Economic analysis of the bio-shielding effect

The comparative simulations indicate that mangroves along selected coasts of Tacloban (Figure 10) could lead to patches of dry sites when Haiyan occurred (Figure 11B). The dry patches correspond to some of the most populous villages of the city. Based on the 2010 population census, the dry patches would have been the sight of 5,500 households. With the data from the household survey, the estimated value attributable to those dry patches is *HV*= Php 2.9 billion (5,500 households x 5 members/household x $2,600 GDP per capita multiplied by Php 40.00/US $1.00 (based on the foreign exchange rate for 2013). The total area of the new mangroves (Figure 10) is about 158 ha, determined through the following considerations. Each cell on the simulated patch is approximately a square with about 36-m length on all sides. The total number of planted cells is 1,216, whereas the total area is 158 ha.

Considering Php100,000 as the minimal spend of fully vegetating a cell, which includes the possible average cost of displacing an existing household, then the total cost of planting is only around Php120 million. The tradeoff between the potential benefit of saved value (Php2.90B) and cost of bio-shielding efforts is staggering—a factor of 24x favoring the benefit. Thus, the rational economic decision would be to situate the restoration efforts along selected areas of the Tacloban coastline.

Discussion

The simulations reveal that mangrove restoration, if done in suitable sites with supportive conditions, can achieve an average tree density of more than 3,000/ha in the medium- (25 years) and long-term (> 50 years). Although different stochastic realizations of the model yield different outcomes, the average results can serve as estimate of the expected long-term outcome. A restoration index is proposed to determine site suitability using data in the first few years of a pilot plantation. Measuring the growth rate, especially the seedling-to-sapling transition, and the sapling and seedling mortality rates may be sufficient to make ξ>1 at least in the first few years post-planting. The initial population pressure suggested by ξ>1 increases the likelihood of seedling establishment toward maturity, enhancing survival. Thus, the value of this index can guide restoration programs in planning for higher chances of success and maximizing the efficient use of resources.

The tree density is an important bio-shield parameter that determines the vegetation-induced water resistance (Carlos et al., 2015). The drag coefficient was the parameter that quantified the resistance of an individual tree to incoming water flow. The tree density accounted for the overall average effect of all mangroves in the area. Even by underestimating the tree density (relative to the average possible from the simulations) in the two sites simulated, Tacloban (4%) and Pan de Azucar (40%), the bio-shield effect is apparent. For Pan de Azucar, the bio-shield effect is substantial even though the mangroves are in an oblique position relative to the Haiyan winds. Due to this orientation, the mangroves did not take the full force of the incoming storm surge. Yet, because of the high tree density, the mangroves sufficiently slowed down the momentum of the reflected water flow. This attenuation of reflected flow is sufficient to reduce the extent of the inundated area inland. For Tacloban, the thinner hypothetical mangroves, which are orientated directly facing the incoming storm surge, still provided a viable level of coastal protection. Drier areas resulting from this protection imply that casualties and costs could have been minimized. The economic analysis based on data from household surveys shows that Tacloban could have saved about US$12 million worth of lost lives and property damage.

Although the study considered only a single genus, the system and methods used may also work for other genera, e.g., *Avicennia* and *Sonneratia* (Carlos et al., 2015). Given that the growth model is found to be robust against species-specific variations, while the vegetation-induced hydrodynamic effects only rely on the physical characteristics of mangroves, the present study should be adaptable to accommodate multi-specific restorations. Detailed field observations and mapping can provide precise information of the mangrove characteristics that are relevant to the hydrodynamics of storm surges. Surveys using LiDAR technology on low-flying unmanned drones (Alon et al., 2019; Marasigan et al., 2019) can provide the precise mapping for quantifying the hydrodynamically relevant vegetation parameters of mangroves.

Conclusion

This prospective simulation study showed an interesting connection between mangrove restoration and coastal protection. The link was obtained by modeling the long-term tree density obtained by applying common restoration practices for coastal mangroves in the Philippines. The tree density was a crucial factor in describing the hydrodynamic drag exerted by mangroves on an incoming water flow, e.g., storm surge, driven by category-5 winds, through short- and long-wave dissipation. While the results showed stable average tree densities of more than 3,000/ha, the stochastic simulations indicate that worse outcomes are possible. Thus, site suitability evaluation is a necessary step toward achieving a higher likelihood of long-term success for the restoration effort. For this step, a restoration index was proposed to quantify the chances of success with field measurements of plant growth and mortality rates that can be performed over the immediate term. Bio-shield simulations showed that even underestimating the tree density of restored mangroves can lead to appreciable levels of coastal protection. A comparison of the actual (no mangrove) and hypothetical (restored mangrove) scenarios revealed that Tacloban could have saved $12 million worth of property damage and lives lost from Haiyan in 2013.

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

DJ formulated the model, performed the simulation experiments, deployed the field surveys, gathered data, interpreted and analyzed the results, and wrote the manuscript.

Funding

Financial support was provided by the Department of Science and Technology with Grant #2014-7373.

Acknowledgments

The author also acknowledges the technical support of the National Research Council of the Philippines; Mr. Joseph Elias Co, for the simulations; Ms. Darahlyn Romualdo, for the field measurements; and the Oscar M. Lopez Center for assistance in the initial draft of the manuscript.

Conflict of interest

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

Publisher's note

Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2022.968420/full#supplementary-material

References

Alon, A. S., Festijo, E. D., and Juanico, D. E. O. (2019). "Tree detection using genus-specific RetinaNet from orthophoto for segmentation access of airborne LiDAR data," in *2019 IEEE 6th international conference on engineering technologies and applied sciences (ICETAS)* (Kuala Lumpur, Malaysia: IEEE), 1–6.

Berger, U., and Hildenbrandt, H. (2000). A new approach to spatially explicit modelling of forest dynamics: spacing, ageing and neighbourhood competition of mangrove trees. *Ecol. Model.* 132 (3), 287–302. doi: 10.1016/S0304-3800(00)00298-2

Carlos, C., Delfino, R. J., Juanico, D. E., David, L., and Lasco, R. (2015). Vegetation resistance and regeneration potential of *Rhizophora, Sonneratia,* and *Avicennia* in the typhoon haiyan-affected mangroves in the Philippines: Implications on rehabilitation practices. *Climate Disaster Dev. J.* 1 (1), 1–8. doi: 10.18783/cddj.v001.i01.a01

Delfino, R. J., Carlos, C. M., David, L. T., Lasco, R. D., and Juanico, D. E. O. (2015). Perceptions of typhoon haiyan-affected communities about the resilience and storm protection function of mangrove ecosystems in Leyte and Eastern samar, Philippines. *Clim. Disaster Dev. J.* 1, 15–24. doi: 10.18783/cddj.v001.i01.a03

Duke, N. C., Meynecke, J. O., Dittmann, S., Ellison, A. M., Anger, K., Berger, U., et al. (2007). A world without mangroves? *Science* 317 (5834), 41–42. doi: 10.1126/science.317.5834.41b

FAO . (2007). *The world's mangroves 1980–2005* Vol. 153 (Rome, Italy: FAO Forestry Paper), 77.

Fulton, M. R. (1993). "Rapid simulations of vegetation stand dynamics with mixed life-forms," in *Vegetation dynamics & global change* (Boston, MA: Springer), 251–271.

Gillespie, D. T. (1976). A general method for numerically simulating the stochastic time evolution of coupled chemical reactions. *J. Comput. Phys.* 22 (4), 403–434. doi: 10.1016/0021-9991(76)90041-3

Holden, W. N., and Marshall, S. J. (2018). "Climate change and typhoons in the Philippines: Extreme weather events in the anthropocene," in *Integrating disaster science and management* (Amsterdam, Netherlands: Elsevier), 407–421.

Kamali, B., and Hashim, R. (2011). Mangrove restoration without planting. *Ecol. Eng.* 37 (2), 387–391. doi: 10.1016/j.ecoleng.2010.11.025

Kamil, E. A., Takaijudin, H., and Hashim, A. M. (2021). Mangroves as coastal bio-shield: a review of mangroves performance in wave attenuation. *Civil Eng. J.* 7 (11), 1964–1981. doi: 10.28991/cej-2021-03091772

Lagmay, A. M. F., Agaton, R. P., Bahala, M. A. C., Briones, J. B. L. T., Cabacaba, K. M. C., Caro, C. V. C., et al. (2015). Devastating storm surges of typhoon haiyan. *Int. J. Disaster Risk Reduction* 11, 1–12. doi: 10.1016/j.ijdrr.2014.10.006

Lee, S. Y., Primavera, J. H., Dahdouh-Guebas, F., McKee, K., Bosire, J. O., Cannicci, S., et al. (2014). Ecological role and services of tropical mangrove ecosystems: a reassessment. *Global Ecol. Biogeography* 23 (7), 726–743. doi: 10.1111/geb.12155

Lewis, R. R.III (2005). Ecological engineering for successful management and restoration of mangrove forests. *Ecol. Eng.* 24 (4), 403–418. doi: 10.1016/j.ecoleng.2004.10.003

Long, J., Napton, D., Giri, C., and Graesser, J. (2014). A mapping and monitoring assessment of the philippines' mangrove forests from 1990 to 2010. *J. Coast. Res.* 30 (2), 260–271. doi: 10.2112/JCOASTRES-D-13-00057.1

Marasigan, R., Festijo, E., and Juanico, D. E. (2019). "Mangrove crown diameter measurement from airborne lidar data using marker-controlled watershed algorithm: Exploring performance," in *2019 IEEE 6th international conference on engineering technologies and applied sciences (ICETAS)* (Kuala Lumpur, Malaysia: IEEE), 1–7.

McIvor, A. L., Spencer, T., Möller, I., and Spalding, M. (2012). Storm surge reduction by mangroves. Natural Coastal Protection Series: Report 2. Cambridge Coastal Research Unit Working Paper 41. Arlington, VA, USA: The Nature Conservancy and Wetlands International.

Morris, M. D. (1991). Factorial sampling plans for preliminary computational experiments. *Technometrics* 33 (2), 161–174. doi: 10.1080/00401706.1991.10484804

Primavera, J. H., and Esteban, J. M. A. (2008). A review of mangrove rehabilitation in the Philippines: successes, failures and future prospects. *Wetlands Ecol. Manage.* 16 (5), 345–358. doi: 10.1007/s11273-008-9101-y

Qiu, Q., Zhou, Z., Lin, J., Zhang, F., Chen, Z., and Yang, X. (2022). Great earthquake and tsunami potential in the eastern makran subduction zone: New insights from geodetic and structural constraints. *Tectonophysics* 837, 229462. doi: 10.1016/j.tecto.2022.229462

Roelvink, D., Van Dongeren, A., McCall, R., Hoonhout, B. M., Van Rooijen, A., Van Geer, P., et al. (2015). XBeach manual. *Deltares Delft Univ. Technology: Delft Netherlands* 1.12: 24–7.

Salmo, S., and Juanico, D. E. (2015). An individual-based model of long-term forest growth and carbon sequestration in planted mangroves under salinity and inundation stresses. *Int. J. Philippine Sci. Technol.* 8 (2), 31–35.

Schaal, B. A., and Leverich, W. J. (1982). Survivorship patterns in an annual plant community. *Oecologia* 54 (2), 149–151. doi: 10.1007/BF00378386

Schmitt, K., Albers, T., Pham, T. T., and Dinh, S. C. (2013). Site-specific and integrated adaptation to climate change in the coastal mangrove zone of soc trang province, Viet nam. *J. Coast. Conserv.* 17 (3), 545–558. doi: 10.1007/s11852-013-0253-4

See, J., and Wilmsen, B. (2022). A multidimensional framework for assessing adaptative justice: a case study of a small island community in the Philippines. *Climatic Change* 170 (1), 1–21. doi: 10.1007/s10584-021-03266-y

Serra, C. S., Martínez-Loriente, S., Gracia, E., Urgeles, R., Gómez de la Peña, L., Maesano, F. E., et al. (2021). Sensitivity of tsunami scenarios to complex fault geometry and heterogeneous slip distribution: Case-studies for SW Iberia and NW Morocco. *J. Geophysical Research: Solid Earth* 126 (10), e2021JB022127. doi: 10.1029/2021JB022127

Su, J., Friess, D. A., and Gasparatos, A. (2021). A meta-analysis of the ecological and economic outcomes of mangrove restoration. *Nat. Commun.* 12 (1), 1–13. doi: 10.1038/s41467-021-25349-1

Suzuki, T., Zijlema, M., Burger, B., meijer, M.C., and Narayan, S. (2012). Wave dissipation by vegetation with layer schematization in SWAN. *Coastal Engineering* 59 (1), 64–71. doi: 10.1016/j.coastaleng.2011.07.006

Temmerman, S., Meire, P., Bouma, T. J., Herman, P. M., Ysebaert, T., and De Vriend, H. J. (2013). Ecosystem-based coastal defence in the face of global change. *Nature* 504 (7478), 79–83. doi: 10.1038/nature12859

Tozer, B., Sandwell, D. T., Smith, W. H., Olson, C., Beale, J. R., and Wessel, P. (2019). Global bathymetry and topography at 15 arc sec: SRTM15+. *Earth Space Sci.* 6 (10), 1847–1864.

Van Kampen, N. G. (1992). *Stochastic processes in physics and chemistry* Vol. Vol. 1 (Amsterdam, Netherlands: Elsevier).

Wang, Y., and Marsooli, R. (2021). Dynamic modeling of sea-level rise impact on coastal flood hazard and vulnerability in New York city's built environment. *Coast. Eng.* 169, 103980. doi: 10.1016/j.coastaleng.2021.103980

Zhang, S. (2013). Haiyan prompts risk research. *Nature* 503 (7476), 324–324. doi: 10.1038/503324a

Zhang, K., Liu, H., Li, Y., Xu, H., Shen, J., Rhome, J., et al. (2012). The role of mangroves in attenuating storm surges. *Estuarine Coast. Shelf Sci.* 102, 11–23. doi: 10.1016/j.ecss.2012.02.021

frontiers | Frontiers in Marine Science

TYPE Systematic Review
PUBLISHED 29 September 2022
DOI 10.3389/fmars.2022.987737

OPEN ACCESS

EDITED BY
Free Espinosa Torre,
Sevilla University, Spain

REVIEWED BY
Alexander Cesar Ferreira,
Federal University of Ceara, Brazil
Mohammad Basyuni,
Universitas Sumatera Utara, Indonesia

*CORRESPONDENCE
Maria Elisa B. Gerona-Daga
mbgerona@up.edu.ph

SPECIALTY SECTION
This article was submitted to Marine Conservation and Sustainability, a section of the journal Frontiers in Marine Science

RECEIVED 06 July 2022
ACCEPTED 05 September 2022
PUBLISHED 29 September 2022

CITATION
Gerona-Daga MEB and Salmo SG III (2022) A systematic review of mangrove restoration studies in Southeast Asia: Challenges and opportunities for the United Nation's Decade on Ecosystem Restoration. *Front. Mar. Sci.* 9:987737. doi: 10.3389/fmars.2022.987737

A systematic review of mangrove restoration studies in Southeast Asia: Challenges and opportunities for the United Nation's Decade on Ecosystem Restoration

Maria Elisa B. Gerona-Daga[1,2]* and Severino G. Salmo III[1]

[1]Institute of Biology, College of Science, University of the Philippines Diliman, Quezon City, Philippines, [2]Division of Natural Sciences and Mathematics, University of the Philippines Visayas Tacloban College, Tacloban City, Philippines

Mangroves provide valuable ecological and socio-economic services. The importance of mangroves is particularly evident in Southeast (SE) Asia where the most extensive and diverse forests are found. To recover degraded mangroves, several SE Asian countries have implemented restoration programs. However, to date, there has been no systematic and quantitative synthesis on mangrove restoration studies in the region. Here, we provide a bibliometric-based analysis of mangrove restoration to provide understanding on trends and future directions needed to meet biodiversity and restoration targets in the region. Following the Preferred Reporting Items for Systematic Reviews and Meta-Analyses (PRISMA) protocol, we analyzed 335 articles (249 articles with ecological attributes; 86 articles with social attributes) published until February 2022 from Scopus and Web of Science databases. Mangrove restoration studies with ecological and social attributes started around the early 1990s mostly from Indonesia, Thailand, Malaysia, Vietnam, and the Philippines. Majority of SE Asian countries have stronger collaboration to western countries rather than within the region. Reasons for restoration vary per country, but mostly were intended to rehabilitate damaged mangroves. Direct planting was the most common restoration method used while hydrological rehabilitation was less practiced. Research on ecological attributes were dominated by biodiversity-related studies focused on flora and fauna, and less on other ecosystem services (e.g., coastal protection, fisheries production, etc.). Studies with social attributes only accounted for <30% of the publications, mostly on topics related to ecological economics. Although mangrove restoration studies are apparent, some thematic restoration foci are needed. We propose priority research topics to help achieve the biodiversity and restoration targets by 2030.

KEYWORDS

mangroves, Southeast Asia (SE Asia), ASEAN network, restoration, ecology, social, policy and governance, bibliometric analysis

Introduction

Mangroves provide a range of ecosystem services including coastline protection (Hochard et al., 2019), carbon storage and sequestration (Donato et al., 2011), and provision of habitat for wildlife and commercially important species (Friess et al., 2020). Mangroves also provide socio-economic benefits like support to livelihood (e.g., ecotourism; Spalding and Parrett, 2019), aqua-silviculture, and forest products (Orchard et al., 2016). Despite these services, reports on mangrove losses at global (Romañach et al., 2018) and regional scales (Richards and Friess, 2016) are apparent.

Southeast Asia (SE Asia) accounts for the world's largest (32.2%; 43,767 km^2) and most diverse mangrove forests (>50 species; Spalding et al., 2010), but unfortunately also has the most extensive mangrove loss (Spalding and Leal, 2021; Bhowmik et al., 2022). Mangrove loss varies regionally, but in many countries the main drivers are the rapid expansion of aquaculture ponds (for fish and shrimp in Vietnam, Indonesia, Thailand, and Myanmar; Luo et al., 2022; for fish in the Philippines; Primavera, 1995), rice production (in Myanmar), and oil palm expansion (in Malaysia and Indonesia; Richards and Friess, 2016). At country-level, Myanmar is the primary mangrove loss hotspot (with 27.6% loss between 2000 and 2014; Estoque et al., 2018) followed by the Philippines (10.5% loss from 1990–2010; Long et al., 2014).

Mangrove losses result in biodiversity lost as well as reduction of ecosystem services (Sannigrahi et al., 2020). Mangroves are regarded as a high-priority ecosystem in a number of international conservation initiatives like the Global Mangrove Alliance (GMA; Bunting et al., 2022). Several international commitments and targets have been set to bolster biodiversity conservation and ecosystem restoration (da Rosa and Marques, 2022), for example, the United Nations Framework Convention on Climate Change (UNFCCC) Paris Agreement, and the Convention on Biological Diversity (CBD) Aichi Targets. The Association of Southeast Asian Nations (ASEAN) member states are signatories to these international commitments (ACB, 2017). While these programs may indicate positive mangrove conservation and restoration strategies, restoration success on the ground is not evenly distributed (Friess et al., 2020) nor systematically reported.

SE Asian countries have been doing mangrove restoration and management for decades. For example, the Matang Mangrove Forest in Peninsular Malaysia was gazetted in 1906 as a permanent forest reserve (Hamdan et al., 2014). In the Philippines, mangrove planting dates back to the 1930s for the supply of construction posts for fish weirs and fuel (Walters, 2003). In Indonesia, mangrove rehabilitation started in the 1930s for timber production (Ilman et al., 2011). In Vietnam, direct planting of fastly-growing *Rhizophora apiculata* was practiced in 1978 on areas affected by the herbicide Agent Orange during the war (Hong, 2001). Clearly, early mangrove rehabilitation practices were focused on establishing mangrove cover for short-term economic gains (i.e., fuel, timber; Suman, 2019). While these practices contributed to the recovery of forest cover, it may no longer be sufficient to address current and future needs (e.g., biodiversity loss, climate-related disturbances, etc.; Andradi-Brown et al., 2013).

Mangrove restoration is a nature-based solution (NbS) advocated to conserve/protect biodiversity and in climate change adaptation and mitigation (CCAM) programs (Zari et al., 2019). However, most restoration programs rarely integrate ecological components (Lewis, 2000) and its social aspects are often neglected (Egan et al., 2011). Despite the proliferation of massive mangrove restoration efforts across SE Asia, a systematic assessment and documentation of its outcomes are still lacking. With different restoration objectives and techniques employed, the general effectiveness of restoration on ecological attributes is not clear (Andradi-Brown et al., 2013) nor whether management efforts are successful or not (Salmo, 2021).

Ecological restoration should aim for substantial ecosystem recovery relative to an appropriate reference model including species composition, community structure, and physical conditions (Gann et al., 2019). For restoration science and practice to advance, it is necessary to learn from previous restoration programs such that failures are minimized, and success is achieved. The experiences in mangrove restoration in SE Asia provide an opportunity to advance mangrove restoration in the region. Hence, in this study, we aim to collate, analyze, and synthesize learnings from mangrove restoration research and identify themes needed to meet the biodiversity and restoration targets in SE Asia.

Methods

We systematically searched on mangrove restoration studies in SE Asia. The term "rehabilitation" is often used interchangeably with "restoration" (Andradi-Brown et al., 2013; Guan et al., 2019). In this context, we used "restoration" as an umbrella term covering a range of intervention activities applied on mangrove forests, including plantation, protection allowing natural regeneration, and habitat restoration (Andradi-Brown et al., 2013). The Preferred Reporting Items for Systematic Reviews and Meta-Analyses (PRISMA) protocol was used for study selection and inclusion (Moher et al., 2009). All analyses were performed after characterization of suitable studies and mapped guidance following the PRISMA 2009 checklist (Supplementary Table 1).

Publications on mangrove restoration studies in SE Asia were identified from Elsevier's Scopus and Clarivate Analytics' Web of Science (WOS) Core Collection databases through two iterative searches. The first search was conducted on October 16, 2021, using the query words "mangrove* AND (restoration OR rehabilitat* OR plantation) AND ("southeast asia" OR Philippine* OR Indonesia OR Malaysia OR Thailand OR Vietnam OR Singapore OR Cambodia OR Myanmar OR Brunei). The detailed query is reported in Supplementary Table 2. Member countries of the Association of Southeast Asian Nations (ASEAN) were specifically added in the query terms to gather researches per individual country. We later updated our datasets and conducted a second search on February 28, 2022, without date restriction to include all relevant publications (Figure 1). The collections from the two databases were merged following Caputo and Kargina (2021). In total, 1,578 publications were retrieved (Scopus: 806; WOS: 772) but only 668 records were retained after duplication removal.

A screening process was conducted based on the selection criteria below:

Criterion 1: We focused on research articles about mangrove restoration with ecological attributes in SE Asia, in general. These included studies from individual ASEAN member countries as well as those involved in more than one country as study sites.

Criterion 2: Articles that described the study sites and how mangrove restoration was done (i.e., direct planting, protection allowing natural regeneration, hydrological rehabilitation, or incorporation of coastal engineering methods). Restoration studies that showed comparison between restored and natural/intact stands as reference sites were also considered.

Criterion 3: Articles classified with social attributes. These included topics on valuation studies, ecotourism potential, ecological economics, environmental education, community engagement, and perception studies.

Criterion 4: Quantitative studies that reflect empirical data on ecological functions. Studies that include assessment metrics related to ecological functions (i.e., biodiversity of flora and fauna, above and below ground biomasses, carbon storage and sequestration, among others) were included. Studies conducted in microcosms, greenhouses, and experimental tanks were excluded.

The first screening involved titles and abstracts for inclusion, resulting to 461 documents considered for full-text screening. In total, 335 articles were included for synthesis (Supplementary Table 3). We further categorized the articles based on the primary objectives for restoration, the restoration approaches used, the ecological attributes assessed, and the social-related attributes reported (Supplementary Table 4). Review articles, editorial materials, conference proceedings, and non-English documents were excluded. Conference proceedings refer to documents with available abstracts only while conference papers are publications with full text articles. Both authors worked independently in the screening and selection of documents for inclusion or exclusion. The extracted data were then validated to check accuracy.

We utilized the Bibliometrix package (Aria and Cuccurullo, 2017) in R studio for bibliographic analysis. Quantitative indices related to scientific productivity, topical trends, and collaboration networks among countries, institutions, and authors were analyzed (Supplementary Table 5). We used the

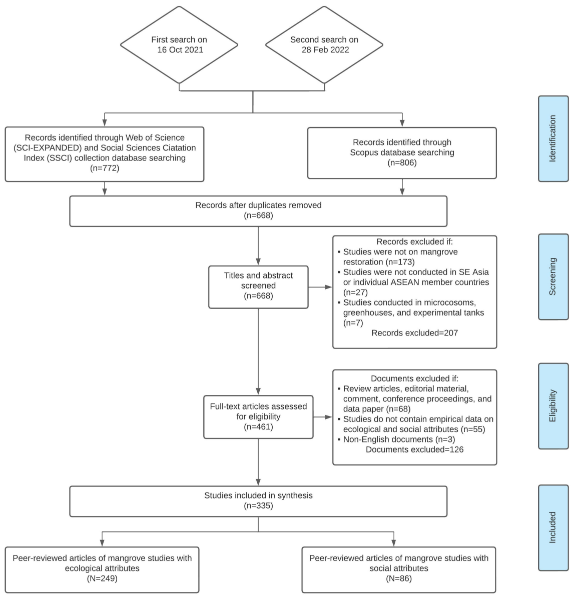


FIGURE 1
PRISMA (Preferred Reporting Items for Systematic Reviews) flow diagram of bibliographic analysis in the systematic review.

web-interface Biblioshiny and data visualization packages from RStudio for the graphical layouts.

Results and discussion

Publication performance and characteristics

Out of 335 total articles compiled, 249 articles (74%) have ecological attributes, and 86 articles (26%) have social attributes. Records on mangrove restoration-related articles in SE Asia started around the 1970s. However, research with ecological and social attributes only appeared in the 1990s (Figure 2). From 1972-2010, restoration records were published at an average of six records per year and greatly increased to 47 per year starting 2011. In 2021, 87 records were published, the highest number of publications per year recorded so far, with topics related to biodiversity (24%), monitoring of land cover changes using remote sensing (17%), and carbon storage and sequestration (14%).

The 249 articles with ecological attributes had an average of 14.2 citation per article and 1.6 citation per article per year. The dataset was composed of articles (212, 85% of the total) and conference papers (37; 15%). Articles with ecological attributes only commenced in 1990 with the work of Martin et al. (1990) being the first and only article recorded in that year. The study investigated the recolonization of *Avicennia* in an oil-polluted mangrove in the east coast of Borneo Is., Indonesia. Parallel analysis on studies with social attributes started in 1993 with the works of Bennett and Reynolds (1993) and Rittibhonbhun et al. (1993). Bennett and Reynolds (1993) investigated the economic and employment values of mangrove forests in Sarawak Mangroves Forest Reserve, Malaysia while the work of Rittibhonbhun et al. (1993) presented the progression of community-based mangrove management and rehabilitation in Trang, Thailand.

The field of ecological restoration (also synonymously with “Restoration Ecology”) was developed during the 1980s (Guan et al., 2019). In SE Asia however, articles related to mangrove restoration were only reflected in the early 1990s, at least from the databases accessed in this study. Some articles may have used different terms other than “restoration” or “rehabilitation” that may underestimate the number of publications reported in this study. The number of articles gradually increased from 1990-2009, then increased to 17 per year since 2010 (Figure 2). Starting 2015, mangrove publications increased at 29% annually with topics related to management approaches (22%), carbon storage (19%), coastal protection (12%), and erosion control and sediment stabilization (9%; Table 1). Topics on greenhouse gas fluxes, species interaction networks, and remote sensing applications emerged in 2017 (Table 1; Figure 7).

A total of 119 different journals published mangrove restoration studies with ecological attributes in SE Asia. The top 20 most relevant journals were dominated by international journals which accounted for 39% of the total, i.e., *IOP Conf. Ser. Earth Environ. Sci.* (7.4%), *Forest Ecology and Management* (3.5%), *Biodiversitas* (3.3%), *Ocean and Coastal Management* (3.3%), and *Estuarine, Coastal, and Shelf Science* (2.9%). Among the 20 most relevant journals, the *Malaysian Forester* (Malaysia) and *Biodiversitas* (Indonesia) were the only country-based journals within SE Asia (Figure 3). Based on Total Citations (TC), *Estuarine, Coastal, and Shelf Science* was the most cited

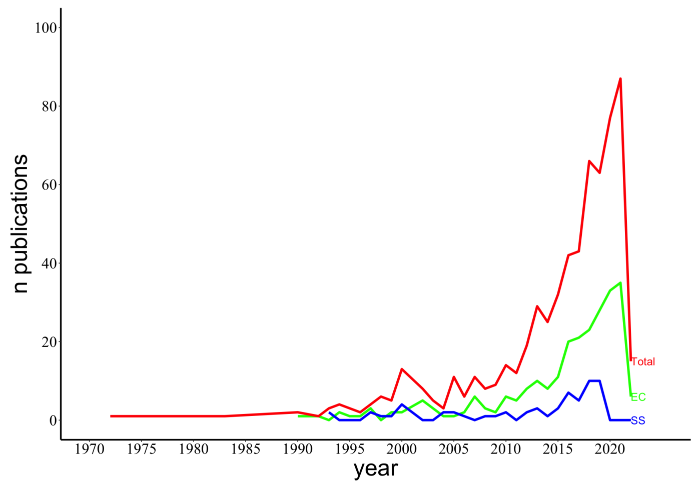


FIGURE 2
Number of documents published annually related to mangrove restoration in SE Asia. Total - all document types; EC, peer-reviewed articles with ecological attributes; SS, peer-reviewed articles with social attributes.

TABLE 1 Most frequent words used in titles, abstracts, and keywords on mangrove studies with ecological attributes in SE Asia.

Title			Abstract			Keywords		
Words	**Occurrences**	**%**	**Words**	**Occurrences**	**%**	**Words**	**Occurrences**	**%**
carbon	34	2.0	species	387	1.5	restoration	32	3.5
restoration	31	1.8	carbon	205	0.8	rehabilitation	24	2.6
coastal	29	1.7	coastal	203	0.8	biodiversity	14	1.5
rehabilitation	29	1.7	natural	201	0.8	forest	11	1.2
diversity	20	1.2	soil	177	0.7	plantation	11	1.2
structure	18	1.1	restoration	168	0.6	biomass	8	0.9
Rhizophora	17	1.0	rehabilitation	164	0.6	blue carbon	8	0.9
ecosystem	16	0.9	ecosystem	149	0.6	climate change	8	0.9
plantation	15	0.9	biomass	132	0.5	diversity	8	0.9
restored	15	0.9	*Rhizophora*	120	0.5	coastal erosion	7	0.8
soil	15	0.9	vegetation	112	0.4	deforestation	7	0.8
community	13	0.8	seedlings	105	0.4	aquaculture	6	0.7
species	13	0.8	diversity	98	0.4	conservation	5	0.5
abandoned	12	0.7	stands	96	0.4	erosion	5	0.5
biomass	12	0.7	planted	95	0.4	remote sensing	5	0.5
coast	12	0.7	structure	94	0.4	sediment	5	0.5
dynamics	12	0.7	density	93	0.4	stand structure	5	0.5
vegetation	11	0.6	plantations	86	0.3	coastal protection	4	0.4
composition	9	0.5	erosion	85	0.3	*Rhizophora*	4	0.4
erosion	9	0.5	management	81	0.3	carbon sequestration	3	0.3

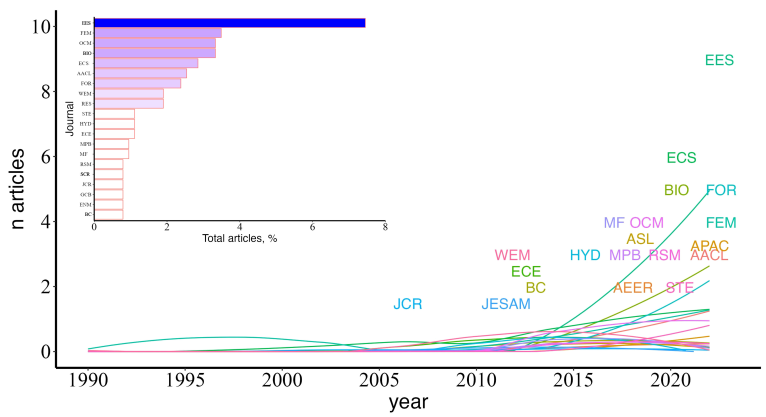


FIGURE 3
Most relevant sources that published mangrove restoration studies with ecological attributes from 1990-2022 (Inset: Most cited sources based on Total Citations). The names of the sources (journals) are placed in years when the highest number of manuscripts was published. Early restoration studies were limited to only five journals but steadily increased after 2015 (> 10 journals). Complete names of sources' letter codes are available in Supplementary Table 6.

journal (>500 citations) followed by *Forest and Ecology Management* (259 citations) and *Journal of Biogeography* (201 citations; Table 2).

The top 20 most relevant documents were dominated by SE Asian-based authors (55%). This indicates a growing number of experts on mangrove restoration in the region. The most relevant document was published in *Ocean and Coastal Management* with 15 citations per year (Lai et al., 2015; Table 2). This work focused on the potential of coastal engineering to mitigate the impact of coastal transformations in Singapore. The study of Giri et al. (2008) published in *Journal of Biogeography* ranked second with 14 citations annually,

TABLE 2 Most frequently cited articles on mangrove studies with ecological attributes in SE Asia and the top cited document per country.

Paper	Author/s and Publication Year	Journal	Citation per year	Total Citation	Country-specific cited documents
The effects of urbanisation on coastal habitats and the potential for ecological engineering: A Singapore case study	Lai et al., 2015	Ocean and Coastal Management	15	121	–
Mangrove forest distributions and dynamics (1975–2005) of the tsunami-affected region of Asia	Giri et al., 2008	Journal of Biogeography	14	203	IDN, THA, MYS
Coastal vegetation structures and their functions in tsunami protection: experience of the recent Indian Ocean tsunami	Tanaka et al., 2007	Landscape and Ecological Engineering	12	196	THA
Coastal erosion and mangrove progradation of Southern Thailand	Thampanya et al., 2006	Estuarine, Coastal and Shelf Science	10	175	THA
Is Matang Mangrove Forest in Malaysia sustainably rejuvenating after more than a century of conservation and harvesting management?	Goessens et al., 2014	PLoS One	8	75	MYS
Carbon stocks in artificially and naturally regenerated mangrove ecosystems in the Mekong Delta	Nam et al., 2016	Wetlands Ecology and Management	8	57	VNM
Mangrove blue carbon stocks and dynamics are controlled by hydrogeomorphic settings and land-use change	Sasmito et al., 2020	Global Change Biology	8	23	IND
Mangrove rehabilitation and intertidal biodiversity: A study in the Ranong Mangrove Ecosystem, Thailand	Macintosh et al., 2002	Estuarine, Coastal and Shelf Science	8	158	THA
Rehabilitating mangrove ecosystem services: A case study on the relative benefits of abandoned pond reversion from Panay Island, Philippines	Duncan et al., 2016	Marine Pollution Bulletin	7	47	PHL
Defining eco-morphodynamic requirements for rehabilitating eroding mangrove-mud coasts	Winterwerp et al., 2013	Wetlands	7	65	IDN, THA, PHL
Mangrove restoration without planting	Kamali and Hashim, 2011	Ecological Engineering	6	76	MYS
The impacts of degradation, deforestation and restoration on mangrove ecosystem carbon stocks across Cambodia	Sharma et al., 2020	Science of The Total Environment	6	19	–
Vegetation and soil characteristics as indicators of restoration trajectories in restored mangroves	Salmo et al., 2013	Hydrobiologia	6	61	PHL
Vegetation regeneration in a sustainably harvested mangrove forest in West Papua, Indonesia	Sillanpää et al., 2017	Forest Ecology and Management	5	35	IDN
Loss and recovery of carbon and nitrogen after mangrove clearing	Adame et al., 2018	Ocean and Coastal Management	6	28	MYS
An integrated approach to coastal rehabilitation: Mangrove restoration in Sungai Haji Dorani, Malaysia	Hashim et al., 2010	Estuarine, Coastal and Shelf Science	6	72	MYS
Community structure dynamics and carbon stock change of rehabilitated mangrove forests in Sulawesi, Indonesia	Cameron et al., 2019a	Ecological Applications	6	22	IDN
Hydroperiod, soil moisture and bioturbation are critical drivers of greenhouse gas fluxes and vary as a function of landuse change in mangroves of Sulawesi, Indonesia	Cameron et al., 2019b	Science of The Total Environment	5	21	IDN
Mangrove forests store high densities of carbon across the tropical urban landscape of Singapore	Friess et al., 2016	Urban Ecosystems	5	34	–
Site-specific and integrated adaptation to climate change in the coastal mangrove zone of Soc Trang Province, Viet Nam	Schmitt et al., 2013	Journal of Coastal Conservation	5	47	–
Carbon sequestration and fluxes of restored mangroves in abandoned aquaculture ponds	Sidik et al., 2019	Journal of the Indian Ocean Region	4	17	IDN

*IDN, Indonesia; VNM, Vietnam; MYS, Malaysia; THA, Thailand; PHL, Philippines.

followed by the work of Tanaka et al. (2007) with 12 citations per year. There were also variations on the most cited documents on a per country basis. For example, the work of Giri et al. (2008) was included in the top 10 most relevant documents for Indonesia, Thailand, and Malaysia. On the other hand, only the work of Nam et al. (2016) appeared for Vietnam. For the Philippines, the most cited sources were the articles of Salmo et al. (2013); Winterwerp et al. (2013), and Duncan et al. (2016). The variations in citation patterns is likely due to the different needs of the country, ecological conditions of the restored sites, or accessibility of the paper. Open access (OA) publications can maximize the benefits of scientific findings for researchers, practitioners, and policy-makers (Iyandemye and Thomas, 2019) resulting in a minimized research-implementation gap in restoration research (Zhang et al., 2018). While positive growth on OA publications have been reported over time, institutional license or publisher's fee is still required for more than 50% of newly-published research (Piwowar et al., 2018). These fees can impede researchers and individuals from low-income countries (Matheka et al., 2014) such as most SE Asian countries to access and publish OA manuscripts.

More than 200 institutions contributed to mangrove restoration studies with ecological attributes. The University of Malaya (UOM) was the most relevant institution in terms of article count (n= 40; Figure 4A). This institution accounted for 16% of the articles, which is approximately double that of the second-ranked institution. The National University of Singapore (NUS), Kasetsart University (KU), the University of Queensland (UQ), and Ateneo de Manila University (ADMU) were the top institutions with 20, 18, 15, and 14 articles, respectively. Among the top 20 most relevant institutions, eight institutions are based outside SE Asia, including Australian and Japanese institutions like UQ, Charles Darwin University (CDU), James Cook University (JCU), and Ehime University. These institutions are regarded as the most productive institutions in terms of mangrove research (Ho and Mukul, 2021).

Over 2,000 authors contributed to mangrove publications in SE Asia. The top five most relevant authors (based on fractionalized article count) were Friess (NUS-Singapore, 8.4), Primavera (ZSL-Philippines, 6.7), Basyuni (USU-Indonesia, 5.6), Salmo (UQ/ADMU/UP-Philippines, 4.0), and Duke (JCU-Australia, 2.7; Figure 4B). Most of the authors included in the list were from SE Asia (65%) and were affiliated with the top 20 most relevant institutions (Figure 4A). Six articles in the top 20 most cited documents (Table 2) were authored by SE Asian authors (Figure 4B) indicating a growing number of experts on mangrove restoration with high scholarly impact.

Based on the country affiliations of corresponding authors, articles were categorized as either single country publications (SCP; reflecting intra-country publication) or multiple country publications (MCP; Figures 5A, B). Malaysia has the highest SCP (62%) while Indonesia, Thailand, Vietnam, and the Philippines have 27 to 44% (Figure 5A). Countries with the highest MCPs were Japan (81%), Australia (93%), Singapore (80%), Philippines (73%), and Vietnam (69%). Thailand and Indonesia have 60% and 56% MCPs, respectively. Among SE Asian countries, Malaysia has the lowest MCP (38%; Figure 5B). The MCPs may indicate the extensive collaboration among countries through research and scholarship grants which provide funding for research, training, and restoration initiatives.

Thematic evolution, topic trends, and collaboration dynamics

The mangrove restoration studies with ecological attributes were dominated by Indonesia (34%), Thailand (16%), Vietnam (16%), Malaysia (15%), and Philippines (13%). Similar pattern was observed in articles with social attributes although the sequence among countries varied: Indonesia (43%), Philippines (20%), Vietnam (15%), Thailand (14%), and Malaysia (4%).

Globally, SE Asia contributes to almost a third of the world's mangrove extent (Spalding and Leal, 2021), with vast covers in Indonesia (2,801,795 ha) and Malaysia (515,743 ha; Bunting et al., 2022). Indonesia is the most productive country in terms of article count (Figures 6A-C). Notably, Myanmar, the third country with highest mangrove cover (496,686 ha; Bunting et al., 2022), has fewer publications (1.6%) over countries with smaller mangrove cover (i.e., Philippines - 260,993 ha, 13%; Thailand - 223,137 ha, 16%; Vietnam - 157,028 ha, 16%; Bunting et al., 2022). The number of published documents per country reflects its importance in a given research field (Guan et al., 2019).

The keyword *plantation* was one of the earliest topics of interest from 2006 until 2019, followed by *deforestation,*

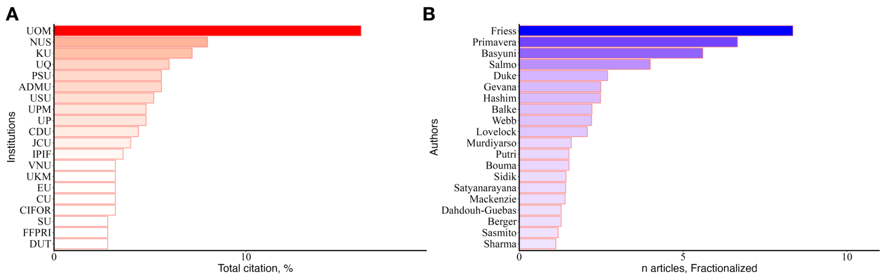


FIGURE 4
Most relevant **(A)** institutions and **(B)** authors in restoration manuscripts in SE Asian mangroves. Most relevant authors were based on fractional authorship which quantifies the individual author's contributions to a published set of papers. Complete names of most relevant institutions are available in Supplementary Table 7.

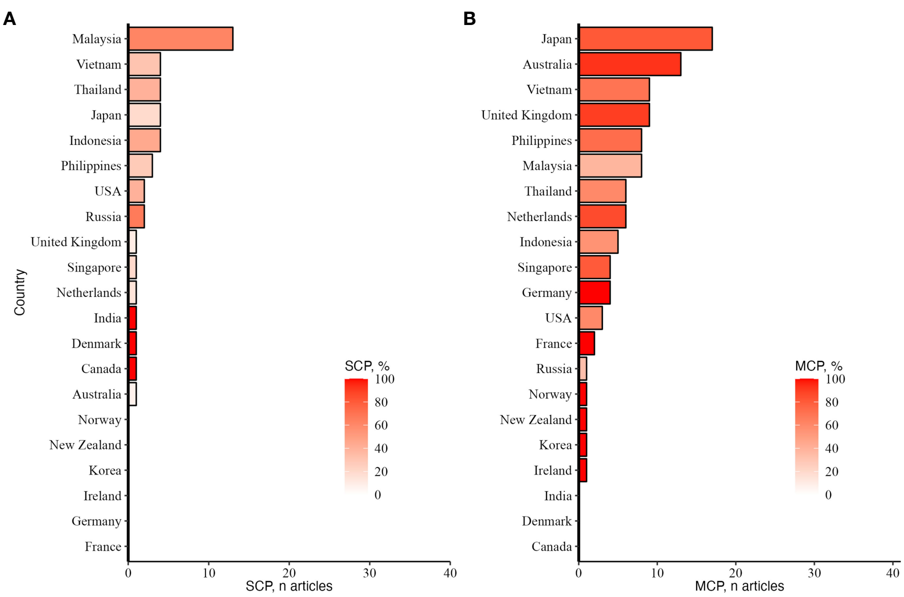


FIGURE 5
Top countries involved in mangrove restoration studies in SE Asia as **(A)** single-country publications (SCP) and **(B)** multiple-country publications (MCP).

rehabilitation, and restoration (Figure 7). Starting 2014, *disturbance*-related topics on erosion and climate change became more frequent. In recent years, more studies used the keywords *blue carbon* and *remote sensing*. Parallel analysis on the most frequent terms associated with titles revealed the words *carbon, restoration, coastal, rehabilitation*, and *diversity* as the most used words (Table 1). This reflects interest in mangrove ecosystem services like coastal protection and carbon storage. With the extreme climatic events (primarily tsunami and typhoons) that affected many countries in SE Asia, protection of mangroves and other coastal vegetation were highlighted (Kathiresan and Rajendran, 2005; Primavera et al., 2016) resulting in the integration of mangrove restoration in coastal rehabilitation plans (Albers and Schmitt, 2015).

The words *natural* and *plantation* were also frequently used in abstracts and keywords (Table 1). Based on the standards of the Society for Ecological Restoration (SER), the use of reference systems (usually referred to as natural mangrove stands;

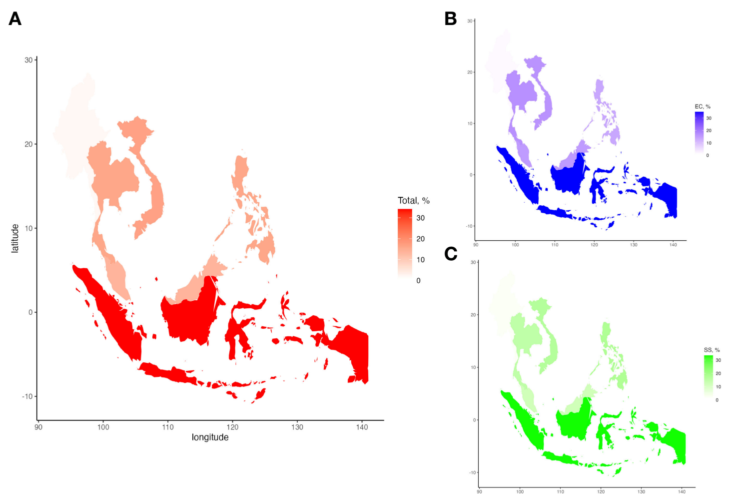


FIGURE 6
Geographical distribution of publications related to mangrove restoration in SE Asia. **(A)** Total - all document types; **(B)**. EC, peer-reviewed articles with ecological attributes; **(C)**. SS, peer-reviewed articles with social attributes

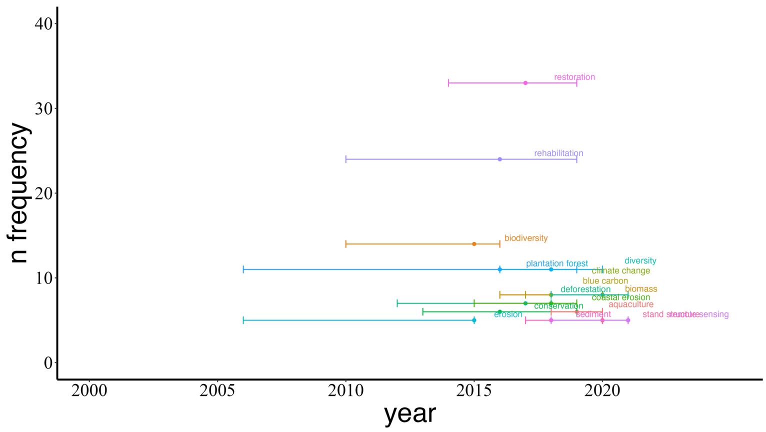


FIGURE 7
Topic trends using author's keywords over the years.

Salmo et al., 2013) in restoration studies, is vital in framing ecological restoration plans and in assessing the success or failure of restoration efforts (Gann et al., 2019). Similarly, analysis on thematic change and evolution using abstracts and titles revealed that topics related to *Rhizophora*, ponds, restoration and seedlings were the foci for the period 1990-2000 (Figure 8). Based on abstract evolution trends, *restoration* studies (1990-2000) evolved to include natural stands (2002-2010) as reference systems. In title evolution trends, the words *rehabilitation* and *plantation* diversified to themes like *diversity, structure*, and *carbon*. Various institutions and government agencies organized mangrove replanting and rehabilitation activities as natural barriers to natural disasters (Barbier, 2007; Baird and Kerr, 2008). Mangroves and other coastal wetlands (i.e., seagrass meadows and tidal salt marshes) are regarded as "blue carbon ecosystems" because of their ability to sequester and store large amounts of carbon (Howard et al., 2017). Salmo and Gianan (2019) reported that disturbances (e.g., catastrophic typhoons) contribute to massive changes in stocks and rates of carbon sequestration. Hydrological alteration in abandoned fishponds was also reported to increase carbon recovery (Matsui et al., 2010).

Research collaboration enables countries with limited experts, experience, and resources to produce impactful studies with other countries (Zhang et al., 2018). Country collaboration networks showed variations of connections between and among SE Asian countries. Among the SE Asian countries, Indonesia (IDN), Malaysia (MYS), Singapore (SGP), Philippines (PHL), Thailand (THA), and Vietnam (VNM) have established networks with the USA, Australia, Netherlands, and to some extent with China. Generally, there are stronger collaborations between SE Asian countries and western countries than among SE Asian countries (Figures 9A-C). In terms of authors' network, SE Asian prolific authors like Friess, Basyuni, Murdiryaso, Primavera, Salmo, and Sasmito have established collaborative networks with other authors (Figure 4B). Similarly, SE Asian research universities (NUS, UP, VNU, Kasetsart, and UOM), non-government organizations (Center for International

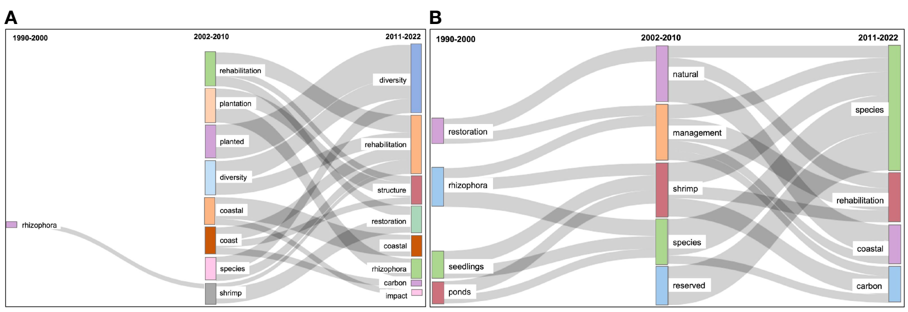


FIGURE 8
Thematic evolution using **(A)** titles and **(B)** abstracts.

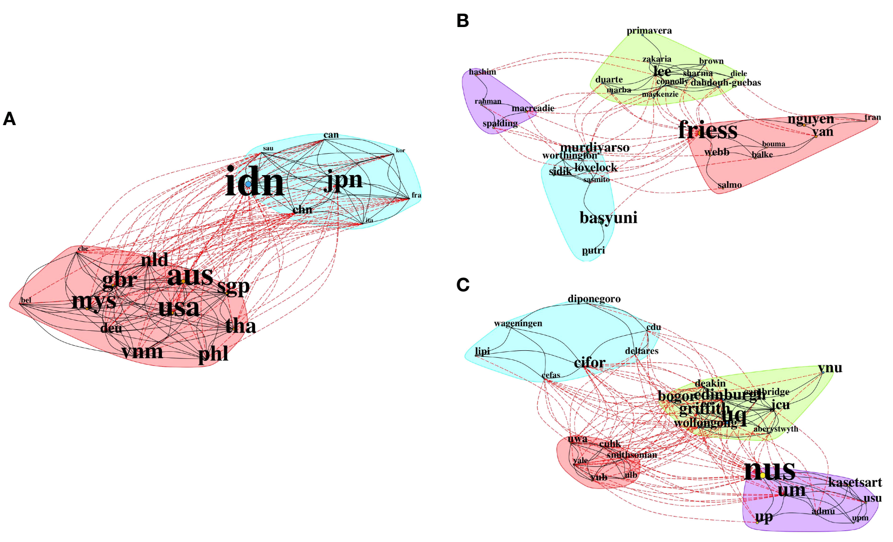


FIGURE 9
Collaboration network by **(A)** countries, **(B)** authors, and **(C)** institutions on mangrove restoration studies with ecological attributes in SE Asia. Country ISO and institutional codes are available in Supplementary Table 8.

Forestry Research [CIFOR], Centre for Environment, Fisheries and Aquaculture (CEFAS), and research institutions like Deltares showed different clusters of networks with western institutions.

Recovery of mangrove areas as drivers for mangrove restoration studies

Threats to mangrove loss (Richards and Friess, 2016) resulting in fragmentation (Bryan-Brown et al., 2020) vary in different SE Asian countries. Coastal lands are high-valued areas for aquaculture, agriculture, settlement, and infrastructure projects for harbors and industries in SE Asia (Slamet et al., 2020). Mangroves, mostly located in coastal fringes, are relatively accessible and always subjected to coastal development pressures (Thakur et al., 2021). Most restoration and rehabilitation programs were implemented to recover mangrove cover. Overall, most of the restoration studies in the region were in response to problems associated with mangrove conversion to aquaculture (58%), coastal erosion (31%), and natural disaster (10%; Table 3). In Indonesia, large-scale conversion of mangroves to aquaculture ponds has been responsible for the destruction of nearly one million ha of mangroves since 1800 (Ilman et al., 2016). Likewise, approximately half of the 279,000 ha of mangroves lost from 1951 to 1988 were converted to aquaculture ponds in the Philippines (Primavera, 2000). The ecological effects of such conversion results in the patchiness of forests affecting biodiversity (Bryan-Brown et al., 2020), carbon storage capacity (Sasmito et al., 2020), and physico-chemical properties of the soil (Matsui et al., 2008), among others. Mangrove forests are highly affected by sediment dynamics. Coastal reclamations for infrastructure, mining, and dam constructions accelerated coastal erosion negatively affecting mangrove ecosystems. Although the issue of natural disasters like tsunami and typhoons have seldom been investigated before the year 2000, such issues are recently getting more studied (10%). The extreme climatic events, such as the Indian Ocean tsunami (in 2004) and Typhoon Haiyan (in 2013), highlighted the importance of restoring coastal vegetation (primarily mangroves) for coastal protection.

Restoration studies in response to mangrove conversion to aquaculture, either for fishpond or shrimp pond production are widespread, with most studies from Indonesia (39%), Philippines (13%), and Thailand (13%). These countries have considerably lost their mangrove cover to aquaculture (Richards and Friess, 2016). Meanwhile, restoration studies in Vietnam were highly focused on coastal erosion reflecting one of the country's main problems. For example, Nguyen et al. (2013) investigated sediment accretion and erosion dynamics through soil particle size fractions in mangrove forests. Notably, only Thailand and Vietnam reported studies on mangrove restoration as a possible solution to pollution (e.g., mining, runoff, etc.). These varying foci of restoration efforts reflect the individual country's local problems and priorities.

Restoration techniques practiced

Three mangrove restoration techniques were commonly reported: direct planting (either monogeneric or multi-species

TABLE 3 Summary of general problems addressed by mangrove restoration from each country and in Southeast Asia.

General problems	Description	BRN	KHM	IDN	LAO	MYS	MMR	PHL	SGP	THA	VNM	SE Asia (Total)
Damage to mangrove habitat	Conversion to aquaculture (fish/shrimp ponds, rice production, oil palm expansion, and herbicide damage)	–	1	64	–	22	4	21	2	28	22	164
Coastal erosion	Eroded floodplain due to rapid reclamation for human settlements and industrial development, etc.	–	2	30	–	15	–	7	2	3	2	61
Natural disaster	Tsunami, typhoons	–	–	11	–	6	1	7	–	5	–	30
Pollution	Poor water and sediment quality due to mining, runoff, etc.	–	-	-	–	–	-	–	–	3	2	5

BRN, Brunei; KHM, Cambodia; IDN, Indonesia, LAO, Laos; MYS, Malaysia; MMR, Myanmar; PHL, Philippines; SGP, Singapore; THA, Thailand; VNM, Vietnam. - no reported article based on the data inclusion criteria in this study.

planting), integration of coastal engineering methods, and hydrological rehabilitation (Table 4). Direct planting, primarily using species from the *Rhizophora* genus, was used as the main restoration technique in all SE Asian countries (74%; Table 4). Monogeneric planting has been widely practiced dating back to the 1930s (Walters, 2003; Ilman et al., 2011) but became more massive and frequent starting in the late 1980s (Primavera and Esteban, 2008; Lee et al., 2019; Arifanti, 2020). Despite the call to follow science-based protocols (i.e., correct site/species matching; Primavera et al., 2016) in mangrove restoration, widespread use of monogeneric *Rhizophora* planting is still reported. In fact, massive restoration programs funded by the national government or in partnership with local government have planted *Rhizophora* in non-mangrove zones (National Greening Program of the Philippines, Primavera et al., 2019) and that post-planting management strategy was based on available funds (Damastuti et al., 2022). Species from the *Rhizophora* genus are widely used planting material due to convenience, easy to collect and plant, and higher survival rate upon initial monitoring (Wodehouse and Rayment, 2019). Hence, the increase of mangrove cover as reported by many countries during the 2nd ASEAN Mangrove Congress in 2017 could be attributed to massive *Rhizophora* planting (Lee et al., 2019). However, the effectiveness of monogeneric planting have been doubted at least in terms of habitat functionality (Barnuevo et al., 2017) and coastal protection (Villamayor et al., 2016) nor in enhancing faunal biodiversity (Salmo et al., 2017; Salmo et al., 2018). Moreover, empirical studies to support its long-term benefits are lacking.

Vietnam along with Indonesia and Malaysia lead in studies on coastal engineering methods, while Indonesia lead in the hydrological rehabilitation methods (Table 4). Hydrological rehabilitation (9%) was advocated prior to planting or to encourage natural regeneration, and some have integrated coastal engineering measures (18%). Studies from Vietnam and Indonesia showed incorporation of engineering measures with various designs to support restoration activities (Albers and Schmitt, 2015; Nguyen, 2018). Different structures and

TABLE 4 Mangrove restoration techniques from each country and in SE Asia.

Restoration techniques	Description	BRN	KHM	IDN	LAO	MYS	MMR	PHL	SGP	THA	VNM	SE Asia (Total)
Direct planting	Monogeneric planting - widely-used species were *Rhizophora apiculata* and *R. stylosa;* Multi-species planting	–	1	63	–	23	4	28	1	31	29	180
Integration of coastal engineering methods	Deployed hard (various types of breakwaters and sea dykes) and soft-engineering methods (T-groins/ fences made up of bamboo, Melaleuca entrapping microsites prior to planting or to encourage natural recruitment)	–	–	9	–	9	–		1	4	11	34
Hydrological rehabilitation	Physical changes made to restore hydrological conditions of the site (considered surface elevation, tidal inundation, etc.) before planting or to encourage natural regeneration	–	–	12	–	1	–		1	2	1	17

BRN, Brunei; KHM, Cambodia; IDN, Indonesia, LAO, Laos; MYS, Malaysia; MMR,Myanmar; PHL, Philippines; SGP, Singapore; THA, Thailand; VNM, Vietnam. - no reported article based on the data inclusion criteria in this study.

construction materials have been tested, including perforated/permeable breakwaters made of bamboo and branches of trees, T-fence, rubble-mound, among others (Akbar et al., 2017; Suripin et al., 2017). In recent years, hydraulic parameters and physical model tests have been incorporated in pre-implementation plans in reducing wave transmission to enhance seedling growth and survival (Le Xuan et al., 2022). A range of hard and soft breakwater structures have been tested to reduce coastal erosion and restore mangrove forests (Thieu Quang and Mai Trong, 2020; Winterwerp et al., 2020; Sartimbul et al., 2021). Successful implementation of breakwaters in Indonesia and Vietnam led to wave energy dissipation (Le Xuan et al., 2022), reduced coastal erosion, sediment build up, and increased colonization rate of mangroves (Akbar et al., 2017; Suripin et al., 2017).

Ecological functions assessed

We identified and categorized nine ecological functions commonly reported in mangrove restoration studies in the region. Floral diversity (34%), carbon sequestration (16%), erosion control and sediment stabilization (14%) were the most commonly reported ecological functions. Other ecological functions related to nutrient cycling (6%), coastal protection (5%), fisheries (5%), and microbial diversity (5%) were relatively less studied (Table 5). The ecological functions reported were probably attempts to link the effectiveness of restored mangroves in delivering ecosystem services (Salmo et al., 2018; Castillo et al., 2022; Comer-Warner et al., 2022). However, documentation and attribution of ecosystem services in restored mangroves are difficult especially if these services are interrelated and there are no baseline datasets to compare with (Salmo, 2021).

Oftentimes, mangrove plant diversity is used as a proxy indicator in the recovery of ecosystem services (Andradi-Brown et al., 2013). The assessment of floral diversity is relatively easier to do (in comparison with other ecosystem services) which could explain why practically all SE Asian countries are reporting it. Flora and fauna diversity, and carbon sequestration characterized most of the studies from Indonesia, while erosion and sediment stabilization, primary productivity, and coastal protection were the primary foci in Vietnam. Despite the relatively fewer studies from Myanmar, they have publications related to carbon sequestration and sediment stabilization. Meanwhile, Malaysia led in microbial diversity assessment (Table 5). The high focus on flora diversity studies can be attributed to the timber value of mangroves. Across SE Asian countries, mangroves are used for posts, and for charcoal and tannin production (Gevaña et al., 2018). Surprisingly however, studies linking restored mangroves with fisheries were seldomly assessed despite the need for food and livelihood of the coastal communities. Among fishery-related topics, nekton communities (e.g., crabs, shrimps, fishes) were the most studied organisms (Salmo et al., 2018; Ridlo et al., 2020; Then et al., 2021) as it is the closest indicative of the food provisioning service of the mangrove ecosystem.

TABLE 5 Ecological functions assessed on mangrove restoration studies from each country and in SE Asia.

Ecological functions	Example of assessment metrics	BRN	KHM	IDN	LAO	MYS	MMR	PHL	SGP	THA	VNM	SE Asia (Total)
Flora diversity	Stand structural characteristics, diversity, distribution, survival and growth patterns	–	1	41	–	12	4	15	2	26	14	115
Carbon sequestration	Above and belowground biomasses, sediment carbon content	–	1	16	–	2	1	7	1	7	6	41
Erosion control and sediment stabilization	Shoreline differences, longshore sediment transport (LST), sedimentation rate, elevation changes	–	–	11	–	9	1	2	–	6	9	38
Fauna diversity	Species composition, richness, diversity and evenness	–	–	20	–	12	–	6	2	5	6	51
Primary productivity	Litter production and accumulation	–	–	6	–	8	–	5	–	3	7	29
Nutrient cycling	Litter decomposition, nutrient load (total nitrogen, available phosphorus)	–	–	3	–	7	–	3	1	4	1	19
Coastal protection	Tide and wave dynamics, wave spectral transformation, wave transmission	–	–	2	–	3	–	1	1	2	5	14
Fisheries and other economically important species	Forest structural characteristics (mangrove stem density, stem diameter, tree height), faunal assemblage patterns, density, abundance	–	–	6	–	2	–	3	–	2	1	14
Microbial diversity	Microbial community composition, distribution	–	–	–	–	3	–	–	1	–	1	5

BRN, Brunei; KHM, Cambodia; IDN, Indonesia, LAO, Laos; MYS, Malaysia; MMR,Myanmar; PHL, Philippines; SGP, Singapore; THA, Thailand; VNM, Vietnam. - no reported article based on the data inclusion criteria in this study.

Recent recognition of the importance of mangroves as a nature-based solution (NbS) have led to various restoration efforts (Cadiz et al., 2020; Basyuni and Simanjutak, 2021; Kusumaningtyas et al., 2022). For example, an increased awareness of the role of mangroves in carbon storage is reflected in our findings (Table 5; see also Tables 1, 2). The inclusion of microbial diversity and other fauna may reflect the recognition of the need to include other important biodiversity components in restoration.

Social attributes assessed

Studies that linked mangrove restoration to social attributes were at least three times lower compared to those that assessed ecological attributes (Figure 2). We identified and classified six categories of social attributes associated with mangrove restoration in Southeast Asia (Table 6). Most of the studies were focused on ecological economics which estimated the economic value of mangroves and its ecosystem services (24%). Topics related to collaboration among different sectors (23%), policies and governance (20%), and community-based restoration (15%) were also explored. Eco-cultural practices (14%) and environmental education (5%) were relatively less studied. Despite the wide range of ecosystem services that the mangroves provide, estimating its non-market ecosystem services results in undervalued estimates of its benefits (Salem and Mercer, 2012). Proper accounting of the multiple services of mangroves is vital for efficient decision-making between conservation and conversion (Song et al., 2021). Collaboration among different sectors (public and private institutions, and community) in implementing restoration projects have been studied for more effective and coordinated conservation efforts (Zhang et al., 2018). For example, local people's participation (Valenzuela et al., 2020) in mangrove restoration with active collaboration of the government and research institutions was an effective strategy towards sustainable and effective mangrove restoration programs (Camacho et al., 2020).

Summary and recommendations for the improvement of mangrove restoration studies in SE Asia

Our study presented a bibliometric-based analysis of mangrove restoration publications in SE Asia to date, providing current knowledge structure, and identifying opportunities for research and collaboration for improved mangrove restoration. We acknowledge that there are a variety of reports and studies (i.e., project technical reports, research studies published in journals not indexed by the databases used) that may not have been covered. However, we argue that the peer-reviewed literature synthesized in this study reflects in general what is available to the wider scientific community. Similar to other bibliometric-studies, data availability and accessibility remains as one of the limitations that may impact the quantification of records and limits the datasets (Mohd Razali et al., 2021). In fact, the research-implementation gap is well documented and criticized in the field of conservation since information from researchers are often not integrated into practice and vice versa (Zhang et al., 2018; Eger et al., 2022). Hence, it is important to make unpublished works be communicated and be subjected to peer-review process in

TABLE 6 Commonly examined social dimensions in mangrove restoration studies from each country and in Southeast Asia.

Social dimensions	Description	BRN	KHM	IDN	LAO	MYS	MMR	PHL	SGP	THA	VNM	SE Asia (Total)
Collaboration among government, NGOs and stakeholders	Explore the role of the different sectors in mangrove restoration and management	–		15	–	–	–	5	–	3	2	25
Ecological economics	Provide estimate of economic value of mangrove ecosystems and their services	–		8	–	3	1	4	–	4	7	27
Community-based restoration	Report of successes and challenges of community-based mangrove restoration	–		10	–	–	–	4	–	2	1	17
Environmental education	Report on the use of mangrove ecosystem as a means to raise awareness of the environment and conservation	–		2	–	2	–	1	–	1	–	6
Ecocultural and practices	Describe local knowledge, practices, and use of mangrove forests	–		8	–	–	–	3	–	2	2	15
Policy and politics	Describe institutional arrangements, issues, policy challenges, and approaches for mangrove conservation	–		11	–	–	–	4	–	2	4	21

BRN, Brunei; KHM, Cambodia; IDN, Indonesia, LAO, Laos; MYS, Malaysia; MMR,Myanmar; PHL, Philippines; SGP, Singapore; THA, Thailand; VNM, Vietnam. - no reported article based on the data inclusion criteria in this study.

order for the wider community to advance restoration practice in the region.

The compiled studies for SE Asia are comparable with the number and rate of publications with other regions (e.g., Asia, Africa) but probably are relatively lower than Australia and the Americas (Ho and Mukul, 2021). There is an increasing trend on mangrove restoration studies with high citations. The commonly cited articles reflect the shared interests among SE Asian countries (particularly on mangrove mapping). Some articles that were authored by researchers from the same country were heavily cited indicating either limited access to international journals or preference to cite locally-published articles. For example, the presence of country-based journals (*Malaysian Forester* for Malaysia and *Biodiversitas* for Indonesia, and to some extent some local journals in the Philippines) could explain high citations in these countries but could also imply limited readership outside the country/region.

The variations of scientific productivity among SE Asian countries is likely due to the differences of resource allocation and the research thrust of the government. Each country may have different investment and strategy in science, fewer research universities and institutions, and less funding opportunities. Mangrove restoration studies are predominantly contributed by Indonesia, Thailand, Malaysia, Vietnam, and the Philippines. These countries have long been doing mangrove research and management and were beneficiaries of several international-funded programs (Salmo, 2019; Hai et al., 2020; Nawari et al., 2021). However, countries like Brunei, Cambodia, Laos, and Myanmar have lower publications despite having considerable mangrove cover and biodiversity (Bunting et al., 2022). The rich history of mangrove research and management in Indonesia (Basyuni et al., 2022), Thailand (Thompson, 2018), Malaysia (Gopalakrishnan et al., 2021), Vietnam (Hai et al., 2020), and Philippines (Garcia et al., 2014) may have contributed to the transition and continuous development of "local experts" in these countries. Clearly, the region benefited from the collaboration networks with authors and institutions outside SE Asia. Almost 50% of the most relevant authors and documents were from outside SE Asia. The region is always at the forefront for international biodiversity conservation and ecosystem restoration programs; hence, it always attracts foreign authors and institutions. For example, the study of Donato et al. (2011) on the contribution of mangroves in abating impacts of climate change revolutionized the "blue carbon" research initially in Indonesia then eventually within and beyond the region. In addition, the long presence of international/regional NGOs and research institutions (e.g., USAID, CIFOR, SEAFDEC, CI, TNC, WWF, etc.) contribute to providing funds to do mangrove research and restoration programs (see for example Figures 4A, 5A, B, 9).

The primary motivation to restore mangroves is to recover mangrove cover (Table 3), mainly through direct planting (Table 4). Although not systematically documented and reported, common indicators for success are usually survival rate and area planted (Kodikara et al., 2017; Wodehouse and Rayment, 2019; Gatt et al., 2022). The reasons for doing mangrove restoration studies are naturally linked to the objectives and practice of restoration. Hence, early topics pursued were on vegetation structure, ecosystem dynamics, composition, biomass and density, deforestation, aquaculture, and management, and recently on "blue carbon", and climate change (Table 1; Figures 7, 8). There are of course common topics among SE Asian countries such as mangrove mapping and coastal dynamics. But there are also topics that are reflective of country-specific needs. For instance, coastal protection is the interest in Vietnam while carbon sequestration along with flora and fauna diversity were the topics pursued in Indonesia. These topics are probably either mandated by the advocacies of the collaborating countries/institutions or a response to international commitments, or both. For example, "blue carbon" is a popular topic probably because it is directly linked to climate change adaptation and mitigation (CCAM) programs with economic and financing opportunities (Chou et al., 2022; da Rosa and Marques, 2022; Macreadie et al., 2022). Interestingly, "biodiversity" which is also a global priority (CBD, 2010; Díaz et al., 2020) is not as comprehensively-studied as "blue carbon", and in fact highly-focused only on measures of vegetation structure related to plant species diversity.

Biodiversity is one of the ecosystem services expected to be recovered in restored mangroves (da Rosa and Marques, 2022). Aside from plant species diversity, other taxa are not assessed/ reported probably because of the inherent difficulties associated with biodiversity studies (e.g., lack of baseline, need to establish gene flow and connectivity, expensive instruments, etc.; Gatt et al., 2022). Aside from biodiversity, other ecosystem functions that are expected to improve following restoration are also not systematically assessed yet (Table 5), probably because there are few biodiversity experts and ecologists that integrate these studies on restored mangroves in the region (but see Basyuni et al., 2021; Then et al., 2021). Outcomes or progress from mangrove restoration programs are needed to document and assess the actual results based on the set objectives. These outcomes are analyzed to show the restoration trajectories over time using sets of restoration indicators (Cadier et al., 2020; Gatt et al., 2022).

Aside from the measured biophysical variables, restoration outcomes should also be assessed on how it contributes to the well-being of the society that are using mangroves and the policy makers that govern mangroves (Bayraktarov et al., 2020; Arifanti et al., 2022b). These outcomes are then integrated to improve mangrove conservation and restoration policies (Lee et al., 2019; Friess et al., 2022; Gatt et al., 2022). The rubrics we adapted in classifying ecological and social attributes are relatively less complete relative to the integrative rubrics or "recovery wheel" proposed by the Society for Ecological Restoration (Gann et al., 2019). However, most (if not all) publications we reviewed here have extremely variable ecological and social variables assessed and therefore limited our ability in evaluating the progress or success of mangrove restoration programs in the region.

Based on the identified gaps and needs, and in line with the international policies/programs, we proposed five priority topics that will enhance the impacts of mangrove restoration studies for SE Asia. We acknowledged that these topics are biased for "biodiversity" and "ecosystem services" simply because these are the pressing needs which we think will highlight the contribution of the region in realizing the targets for the UN's Decade on Ecosystem Restoration in 2030. Although some of these topics may be considered an independent topic on its own, these are complementary to each other. It is also possible that there are topics that are equally important that we may have unintentionally excluded.

Restoration areas and methods

The region already has lessons and experiences (some of it are even painful) on mangrove restoration programs (Primavera et al., 2011; Gevaña et al., 2018; Lee et al., 2019). The massive "planting" programs implemented in the 1990s provided learnings on what system will work as opposed to programs that are bound to fail (Barnuevo et al., 2017; Wodehouse and Rayment, 2019). In general, restoration programs that are implemented at smaller/local-scales and in mid to upper intertidal areas have a higher chance to succeed as opposed to massive planting in lower intertidal coastal fringes. Although restoration at a smaller scale has a higher chance to succeed, it has to be balanced with the urgency of the need to recover mangrove areas. A set of criteria has to be defined to delineate and prioritize restoration areas. There are already existing rubrics in site selection and prioritization, for example former mangrove areas, proximity to existing intact/healthy mangroves, tidal range, and projected vulnerability to sea-level rise, among others (Primavera et al., 2012; Lewis et al., 2019; Teutli-Hernández et al., 2020). We proposed to add mangrove plant diversity based on historical species composition/distribution to integrate data on genetic connectivity for transboundary biodiversity conservation (as posited by Wee et al., 2019). Suitable mangrove species at specific sites can be then determined similar to the proposition of Su et al. (2022). Worthington and Spalding (2018) estimated ca. 3,000 km^2 potential restorable areas in SE Asia. This estimate needs to be further calibrated at country-specific (or even local/site-specific) levels following set rubrics to come up with a reasonable target area. Hopefully, the estimated target areas could match the projected needs of increasing mangrove cover by at least 20% in 2030 (GMA; https://mangrovealliance.org/). A significant restorable area would be the abandoned aquaculture fish/shrimp ponds which account for ca. 23,000 km^2 in the region (Luo et al., 2022). We acknowledge that delineating or even prioritizing areas for restoration will be challenging as these are the same areas where the government institutions are also considering as human settlement and reclamation areas for coastal development (Powell, 2021; Tinh et al., 2022). We argue however that addressing this challenge will be more beneficial in the long run as it will come up with a more realistic plan.

Mangrove restoration in climate change adaptation and mitigation programs

For a long time, the primary driver for doing mangrove restoration in the region is to recover mangrove forest (Tables 3, 4). Hence, the choice of species and planting sites were deliberately set for fast-growing species and/or in areas that can be easily restored. However, SE Asia (and as part of Asia-Pacific region) is considered as the most vulnerable region (Noor and Abdul Maulud, 2022) against climate change-related disasters (e.g., typhoons, tsunamis, rising sea-level, etc.). Conventional restoration objectives and designs particularly monogeneric planting in coastal fringes will no longer be sufficient to meet the challenges and complexities needed to adapt (and/or mitigate) the impacts of climate change. Mangroves are commonly advocated as a NbS (Jordan and Fröhle, 2022) indicating mangroves can naturally recover and could even expand in inland areas through natural re/colonization (Winterwerp et al., 2020). The natural recolonization, although relatively "free", is estimated to take a minimum of 10 to 25 years (Salmo et al., 2013) to come up with a developed forest, a period that is too long to wait to be adaptive to climate change impacts. The objectives and designs of restoration programs will have to be modified to be more strategic to adapt to the impacts of climate change. Recent integration of innovative (e.g., bamboo, *Melaleuca* entrapping microsites, rubble-mounds) and technological designs (e.g., coastal engineering) needs to be expounded (Table 4) in more areas. We acknowledge that technological innovations (permeable dams, dykes, and T-groins/fences) will entail substantial cost, a proposition that may be financially difficult for most SE Asian countries. We argue however that technological innovation is not just an option but in fact is a necessity to ensure faster and sustained mangrove forest recovery. For instance, an optimized dyke design considering hydrodynamic loads, including water levels successfully facilitated restoration of mangrove areas in Vietnam (Albers and Schmitt, 2015). Similarly, permeable dams constructed at various locations in Indonesia helped rehabilitate mangrove areas through re-establishment of sediments (following the Build with Nature approach; Winterwerp et al., 2020). A hybrid of mangrove protection, natural recolonization and technological-innovation can also be adapted in anticipation for the increased urgency for mangroves to adapt to uncertain climate change-induced conditions (e.g., less precipitation, rising sea levels, and extreme weather events; Friess et al., 2022). If properly done, the region will be poised to demonstrate the effectiveness of mangrove restoration programs in adapting and mitigating the impacts of climate change particularly on sequestering carbon, reducing GHG emission, and increasing surface elevation.

Monitoring recoveries of biodiversity and ecosystem services

The lack of monitoring data in most mangrove restoration projects has been a perennial problem in the region. Monitoring reports, if available, are confined to short-term monitoring that can potentially misinterpret the success or failure of a restoration project. Biodiversity-related studies on restored mangroves are already notable in the region, although limited to floral measures. Conversely, faunal species and ecological functions and services are rarely reported (Table 5) nor its relationship with the restored mangrove vegetation. Faunal studies have been more focused on molluscs (gastropods, bivalves, and crustaceans) probably as it provides direct food for the nearby coastal communities (Table 5). When mangroves mature, its vegetation structures (e.g., density, biomass, canopy, etc.) and structural complexity are expected to show progression over time following chronosequence (Salmo et al., 2013; Salmo et al., 2017; Salmo et al., 2018). At each forest development stage, the changes in vegetation will improve sediment properties which are then expected to attract different faunal cohorts, and probably also with different trophic levels. The shifts in species composition and dominant species at different forest developmental stages are important to assess linkages between restored mangroves and faunal composition/biodiversity. Similarly, such linkages can be used to establish restoration indicators and eventually be used to infer progress/success (Salmo et al., 2017; Barbanera et al., 2022). Beyond mangrove ecosystems, there is a need to expand and consider connectivity studies and include equally important but less studied taxa (i.e., microorganisms, wildlife fauna). Migration patterns of species (i.e., migratory shorebirds) and interconnectivity of adjacent habitats (coral reef, seagrass, and mangroves) are rarely studied. For example, the health of adjacent ecosystems may also play a role in the health of restored mangroves (see for example Sharma et al., 2017). Likewise, knowledge on mangrove biodiversity should be properly documented and systematically organized. Effective use of biodiversity data requires integration of disconnected datasets (Heberling et al., 2021) for strategic prioritization. We suggest the use of a database as a repository of biodiversity-related information. In this manner, information will be collated (at country-level) and integrated at a regional-level to provide timely and relevant information to researchers and policy makers.

There is a general consensus on accelerated global biodiversity loss in most ecosystems but could be higher in mangroves (Polidoro et al., 2010; Hughes, 2017). Southeast Asia is known as a biodiversity hotspot (Hughes, 2017), although evidence on patterns and rates of biodiversity losses in mangroves are lacking (Sodhi et al., 2010; Tan et al., 2022). One of the primary drivers for restoration is to complement biodiversity conservation (da Rosa and Marques, 2022). To meet agendas for biodiversity conservation and mitigation of climate change, we proposed that vegetation metrics be correlated/related to ecosystem services and functions. We acknowledge that vegetation metrics are relatively easier to measure and reflect the traits that recover faster (Cadier et al., 2020; Gatt et al., 2022). However, relating these metrics to ecological functioning (e.g., habitat provisioning for biodiversity) will be more strategic to quantify the effectiveness (or ineffectiveness) of restoration (Ulfa et al., 2018; Barbanera et al., 2022). Aside from vegetation metrics, we also recommend the comparative assessment on biodiversity and ecosystem services among intact, disturbed, and restored mangroves to provide information on restoration trajectory patterns, including species that are effectively restored. Aquaculture is considered as one of the main drivers of mangrove loss in the region (Richards and Friess, 2016; Gandhi and Jones, 2019), which has expanded about 2.5 times for the last 25 years (Luo et al., 2022). With changing policy discourse surrounding the utilization and value of mangroves, the massive clearing of mangroves for aquaculture from 1950s to 1980s (Primavera, 2000; Valiela et al., 2001) have transformed to mangrove reforestation since 2011 (Song et al., 2021; Arifanti et al., 2022b). However, a substantial area of abandoned, undeveloped and underutilized (AUU) ponds are still to be restored (Primavera et al., 2011; delos Santos et al., 2022). The existence of AUU ponds in the region provides a rare opportunity to assess biodiversity/ecosystem services and its recovery patterns from a damaged mangroves to a supposedly "healthy" mangroves.

We acknowledge that conventional biodiversity monitoring methods (e.g., transect, plot, capture-based samplings, etc.) are still important in providing empirical datasets. However, these methods are time-consuming and expensive (Taddeo and Dronova, 2018). The urgency to document and assess biodiversity calls for revolutionary monitoring methods such as the use of environmental DNA (eDNA) which can supplement conventional biodiversity monitoring methods (see for example Oka et al., 2021; Polanco Fernández et al., 2021). This technological advancement provides a new avenue on monitoring biota which is a non-destructive and rapid method. Another tool that has been progressively integrated in monitoring changes in mangroves and vegetation dynamics is remote sensing. Free access to satellite imagery can potentially support consistent assessment and monitoring of spatio-temporal changes of mangrove forests at a lower cost (Alexandris et al., 2013), yet provide reliable tracking on restoration progress (Reddy, 2021). This is particularly useful in challenging field conditions and difficult-to-access areas like mangrove ecosystems. However, monitoring might be challenging for small-scale restorations with low resolution imagery.

Policies, governance, and community engagement

Policies that are related to mangrove restoration already exist in the region, mostly adapting global policies that aims to conserve biodiversity (e.g., CBD Aichi Targets), to recover ecosystem services (e.g., SDG), and to reduce GHG emission (e.g., UNFCCC Paris Agreement), among others. The scope and context varied widely among SE Asian countries. The need to upscale and accelerate mangrove restoration will need realignment of the existing policies and synergies across institutions (Mursyid et al., 2021) to include financing, investments, and clear objectives through an overarching organization (e.g., UNFCCC; Waltham et al., 2020; Bhowmik et al., 2022). More importantly, the policies should consider mangrove restoration as part of the national development agenda (Mursyid et al., 2021; Arifanti et al., 2022a) that is integrated in the local coastal management plans (Quevedo et al., 2021a). If all AUU ponds are restored, there is a huge potential to contribute to each country's nationally determined contribution (NDC) targets. To date, Indonesia (Mursyid et al., 2021) and the Philippines (Salmo et al., 2021) have drafted mangrove roadmaps. The realigned policies will need to be "ambitious" following science-based and evidence-based protocols (*sensu* Friess et al., 2022). Fortunately, the ASEAN is already available which could be tapped to facilitate the development of common mangrove policies across countries (Palis et al., 2014; Arifanti et al., 2022b).

One of the priority policy needs is to ensure that the remaining mangroves will be effectively conserved (Lee et al., 2019) and to prevent activities that will damage the mangroves (see also example of coastal reclamation project in Jakarta Bay; Slamet et al., 2020). At the least, coastal development plans should integrate protection of mangroves rather than subjecting it to land reclamation activities. Moreover, a policy on science-based green-gray coastal engineering is critical to adapt to changing climatic conditions (Bruins et al., 2019). Complementary to mangrove protection is an enabling policy that will institutionalize upscaled and accelerated mangrove restoration programs in priority areas. These programs will need funding which could be beyond the capacity of most countries (Buchner et al., 2019; Ong, 2021). Some ASEAN countries are already beneficiaries of donor-assisted mangrove management programs (see for example Quevedo et al., 2021b). However, the realigned policies will need to provide supplemental funding and attract investments through public-private partnership or through the corporate social responsibility (CSR) program (Asaeda et al., 2013; Amin et al., 2021). The benefits derived from restored mangroves should be recognized distinct from the conserved mangroves (Ellison et al., 2020). Some coastal communities demonstrated their effectiveness in managing the restored mangroves (see for example Panay Island, Cogtong Bay, and Quezon in the Philippines; Katon and Pomeroy, 2000; Thompson et al., 2017; Gevaña et al., 2018; and in Indonesia, Basyuni et al., 2022). Their contribution to restoration should be recognized and incentivized either through monetary rewards or tenurial instruments (Lovelock and Brown, 2019) to encourage community engagement (Quevedo et al., 2020).

Mangrove conservation and restoration both contribute to climate change adaptation, hence restoration offers opportunities to develop market-based mechanisms in offsetting carbon emissions (Macreadie et al., 2022). For instance, carbon credits generated from planting 18 million trees by the communities in Indonesia is being used to repay project costs (Herr et al., 2019). These monetary-based mechanisms can potentially pay restoration project costs and support local communities through livelihood projects. The communities, as beneficiaries, will then serve as stewards in managing the restoration projects. However, despite the potential contribution of carbon credits to improve local livelihoods, many challenges remain. For one, the perceived social benefits (e.g., increased food and income) from restored mangroves, in general, and from carbon credits, in particular, may take time before the communities can realize it. While waiting for the tangible benefits, the communities have the tendency to resort to illegal activities (e.g., mangrove cutting) in pursuit of immediate and short-term economic gains (see for example Ken et al., 2020). Long-term growth and recovery of mangrove forests should be given more emphasis rather than the hype on carbon offsets (Wernick and Kauppi, 2022). Ensuring an enabling policy environment, including institutionalized funding mechanisms that will incentivize communities (Ken et al., 2020), is critical in achieving long-term restoration goals. Price-based instruments (e.g., tax credits, carbon credits; Lee et al., 2019) should be incorporated in the policy to incentivize coastal stakeholders managing the restored mangroves (Song et al., 2021; Macreadie et al., 2022) but should de-incentivize activities that damage mangroves (e.g., taxes on deforestation; Lee et al., 2019; Lin et al., 2022).

Policies supporting research and systematic monitoring of restoration programs are needed (Maina et al., 2021). The lack of monitoring data in most mangrove restoration projects has been a perennial problem in the region. Empirical studies using standardized restoration tracking tools (see for example Gatt et al., 2022) that document and assess both successes and failures of restoration programs are needed to provide timely inputs to mangrove managers (Friess et al., 2022). Restoration outcomes should be properly stored in a mutually agreed knowledge repository site similar to the biodiversity monitoring platform of the ASEAN Center for Biodiversity (ACB).

Strengthening of ASEAN network

Our study has shown considerable networks among authors, institutions, and countries among SE Asian nations although collaborations with the more developed western countries are more apparent (Figure 9). Establishment of relationships between individuals, institutions, and countries can facilitate the formation of common goals and concerted restoration efforts. Collaboration can offer a range of benefits through knowledge and resource sharing, and cooperative problem solving. The network might also be tapped to enhance mangrove research and management in other SE Asian countries currently with limited research. While external collaborations are helpful, we argue that enhancing collaboration among SE Asian countries will strengthen the network and could probably be more sustainable. The ASEAN can be tapped as a general platform to enhance the network, particularly on sharing best practices, in developing common mangrove management guidelines, in developing collaborative research, and in sharing the state of the environment report (ACB, 2017). To minimize the research-implementation gap, it is necessary that research be communicated in a wider platform. Equally important step is the peer-review process to publish high-quality research articles that can meaningfully contribute to restoration practice. To realize this, we proposed an ASEAN journal focused on mangrove restoration, conservation, and management with a multinational scientific editorial board who are experts in mangrove studies. This can potentially increase the readership of ASEAN-based mangrove studies beyond the region.

For mangroves in particular, to date, there were two scientific fora on ASEAN Mangrove Congress (held in 2012 and 2017 in the Philippines; Palis et al., 2014; Lee et al., 2019). The congress aimed to strengthen mangrove research and development in the ASEAN region through the enhancement of inter-agency and inter-sectoral coordination at the national and regional levels. Priority research areas and policy gaps were identified which were later on adopted through a resolution. Some of these resolutions include the establishment of a common database (e.g., mangrove information center), conduct of conservation and restoration programs, and institutionalization of a mechanism linking mangrove science, policy, and action. The ASEAN Mangrove Congress was initially planned to be held every three years where hosting will be on a rotational basis (Palis et al., 2014), however the 2nd Congress was made possible after five years (Lee et al., 2019). We recognize the inherent challenges that each individual country and the entire ASEAN might encounter (primarily on funding and administration) not to mention the sustainability of such an initiative. However, if only ASEAN members commit and recognize the importance of ASEAN cooperation in addressing regional mangrove research and management initiatives, then an ASEAN Mangrove Congress can be pursued on a bi-annual basis, parallel with the ASEAN Summit.

Final remarks

As mangrove restoration initiatives grow owing to its recognition as a "blue carbon ecosystem", so will the need for mangrove restoration studies. While our findings represent the current status of mangrove restoration studies in SE Asia, we acknowledge that the field of "restoration ecology" is still developing. The inclusion of social attributes, in addition to the classical ecological attributes assessment in mangrove restoration can potentially enhance restoration outcomes. Integration of social dimensions in ecological restoration of mangroves can increase the socio-cultural value of mangroves and at the same time increase scientific output through community engagement (or through "citizen science", i.e., mapping mangroves with local community partners, local knowledge, practices, and use of mangrove forests). Future restoration strategies may benefit to focus on citizen science, and include social attributes, in addition to the usual focus of ecological attributes in mangrove restoration. Regional stakeholders' collaboration, including integration of science-based methods into practice, and improved communication across sectors, will significantly contribute to knowledge transfer. Research topics suggested in this study provide a path forward to improve mangrove restoration, and aid in the development of national and international restoration and conservation strategies, and eventually to contribute to the United Nation's Decade on Ecosystem Restoration.

Data availability statement

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

Author contributions

MG-D: conceptualization, methodology, formal analysis, investigation, data curation, writing – original draft, review, and editing. SS: conceptualization, data visualization, writing – review and editing, supervision. All authors are responsible for and agreed to the publishing of the final version of the manuscript.

Funding

MG-D received a scholarship grant for doctoral fellowship from DOST-ASTHRDP (Department of Science and

Technology-Accelerated Science and Technology Human Resource Development Program). SS contribution was partially supported by Partnerships for Enhanced Engagement in Research (PEER) under USAID cooperative agreement number AID-OAA-A-11-00012. The funders had no role in study design, data collection and analysis, decision to publish, or preparation of the manuscript.

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.

Publisher's note

All claims expressed in this article are solely those of the authors and do not necessarily represent those of their affiliated organizations, or those of the publisher, the editors and the reviewers. Any product that may be evaluated in this article, or claim that may be made by its manufacturer, is not guaranteed or endorsed by the publisher.

Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2022.987737/full#supplementary-material

References

ACB (2017). *ASEAN centre for biodiversity (ACB) ASEAN biodiversity outlook 2* (Laguna, Philippines: ASEAN Centre for Biodiversity, Philippines). Available at: https://asean.chm-cbd.net/sites/acb/files/2020-03/ASEAN%20Biodiversity%20Outlook%202.pdf

Adame, M. F., Zakaria, R. M., Fry, B., Chong, V. C., Then, Y. H. A., Brown, C. J., et al. (2018). Loss and recovery of carbon and nitrogen after mangrove clearing. *Ocean Coast. Manag.* 161, 117–126. doi: 10.1016/j.ocecoaman.2018.04.019

Akbar, A. A., Sartohadi, J., Djohan, T. S., and Ritohardoyo, S. (2017). The role of breakwaters on the rehabilitation of coastal and mangrove forests in West kalimantan, Indonesia. *Ocean Coast. Manag.* 138, 50–59. doi: 10.1016/j.ocecoaman.2017.01.004

Albers, T., and Schmitt, K. (2015). Dyke design, floodplain restoration and mangrove co-management as parts of an area coastal protection strategy for the mud coasts of the Mekong delta, Vietnam. *Wetl. Ecol. Manage.* 23, 991–1004. doi: 10.1007/S11273-015-9441-3/FIGURES/6

Alexandris, N., Chatenoux, B., Harriman, L., Lopez Torres, L., and Peduzzi, P. (2013). Monitoring mangroves restoration from space. *Geneva: UNEP/GRID.* Available at: https://www.unep.org/resources/report/monitoring-restoration-mangrove-ecosystems-space-0

Amin, M. A., Mulyana, D., Damar, A., and Budiman, M. A. K. (2021). Effectiveness and impact studies of mangrove rehabilitation in the northern coast of West Java: A case study in karawang regency. *IOP Conf. Ser. Earth Environ. Sci.* 744, 12002. doi: 10.1088/1755-1315/744/1/012002

Andradi-Brown, D. A., Howe, C., Mace, G. M., and Knight, A. T. (2013). Do mangrove forest restoration or rehabilitation activities return biodiversity to pre-impact levels? *Environ. Evid.* 2, 1–8. doi: 10.1186/2047-2382-2-20/TABLES/2

Aria, M., and Cuccurullo, C. (2017). Bibliometrix: An r-tool for comprehensive science mapping analysis. *J. Informetr.* 11, 959–975. doi: 10.1016/J.JOI.2017.08.007

Arifanti, V. B. (2020). Mangrove management and climate change: A review in Indonesia. *IOP Conf. Ser. Earth Environ. Sci.* 487, 12022. doi: 10.1088/1755-1315/487/1/012022

Arifanti, V. B., Kauffman, J. B., Subarno, ., Ilman, M., Tosiani, A., and Novita, N. (2022a). Contributions of mangrove conservation and restoration to climate change mitigation in Indonesia. *Glob. Change Biol.* 28, 4523–4538. doi: 10.1111/gcb.16216

Arifanti, V. B., Sidik, F., Mulyanto, B., Susilowati, A., Wahyuni, T., Subarno, S., et al. (2022b). Challenges and strategies for sustainable mangrove management in Indonesia: A review. *Forests* 13, 695. doi: 10.3390/f13050695

Asaeda, T., Tsuneizumi, E., Kanehira, H., and Kanesaka, Y. (2013). "Streamlining corporate responsibility into effective mangrove rehabilitation and management scheme in the Philippines," in *Proceedings of regional symposium on mangrove ecosystem management in southeast Asia: Mainstreaming mangroves.* Eds. B. Santoso and T. Kusano (Surabaya, Indonesia: Indonesian Ministry of Forestry), 274–281.

Baird, A. H., and Kerr, A. M. (2008). Landscape analysis and tsunami damage in aceh: Comment on iverson and prasad, (2007). *Landsc. Ecol.* 23, 3–5. doi: 10.1007/S10980-007-9152-0

Barbanera, A., Markesteijn, L., Kairo, J., Juma, G. A., Karythis, S., and Skov, M. W. (2022). Functional responses of mangrove fauna to forest degradation. *Mar. Freshw. Res.* 73, 762–773. doi: 10.1071/MF21257

Barbier, E. B. (2007). In the wake of tsunami: Lessons learned from the household decision to replant mangroves in Thailand. *Resour. Energy Econ.* 30, 229–249. doi: 10.1016/j.reseneeco.2007.08.002

Barnuevo, A., Asaeda, T., Sanjaya, K., Kanesaka, Y., and Fortes, M. (2017). Drawbacks of mangrove rehabilitation schemes: Lessons learned from the large-scale mangrove plantations. *Estuar. Coast. Shelf Sci.* 198, 432–437. doi: 10.1016/j.ecss.2017.02.015

Basyuni, M., Sasmito, S. D., Analuddin, K., Ulqodry, T. Z., Saragi-Sasmito, M. F., Eddy, S., et al. (2022). "Mangrove biodiversity, conservation and roles for livelihoods in Indonesia," in *Mangroves: Biodiversity, livelihoods and conservation* (Singapore: Springer Nature Singapore), 397–445. doi: 10.1007/978-981-19-0519-3_16

Basyuni, M., and Simanjutak, E. O. (2021). Species composition and carbon stock estimation in pulau sembilan secondary mangrove forests, north Sumatra, Indonesia. *IOP Conf. Ser. Earth Environ. Sci.* 713, 12014. doi: 10.1088/1755-1315/713/1/012014

Basyuni, M., Slamet, B., Sulistiyono, N., Munir, E., Vovides, A. G., and Bunting, P. (2021). Physicochemical characteristic, nutrient, and fish production in different types of mangrove forest in north Sumatra and aceh provinces of Indonesia. *Kuwait J. Sci.* 48, 1–14. doi: 10.48129/kjs.v48i3.9160

Bayraktarov, E., Brisbane, S., Hagger, V., Smith, C. S., Wilson, K. A., Lovelock, C. E., et al. (2020). Priorities and motivations of marine coastal restoration research. *Front. Mar. Sci.* 7. doi: 10.3389/fmars.2020.00484

Bennett, E. L., and Reynolds, C. J. (1993). The value of a mangrove area in Sarawak. *Biodivers. Conserv.* 2, 359–375. doi: 10.1007/BF00114040

Bhowmik, A. K., Padmanaban, R., Cabral, P., and Romeiras, M. M. (2022). Global mangrove deforestation and its interacting social-ecological drivers: A systematic review and synthesis. *Sustainability.* 14, 4433. doi: 10.3390/su14084433

Bruins, J., Corwin, E., Pangilinan, J., Pidgeon, E., Taylor, S., and Ng, K. (2019). "Building coastal resilience for disaster risk reduction and climate change adaptation through green-gray infrastructure," in *International conference on sustainable infrastructure 2019* (Reston, VA: American Society of Civil Engineers), 78–88. doi: 10.1061/9780784482650.009

Bryan-Brown, D. N., Connolly, R. M., Richards, D. R., Adame, F., Friess, D. A., and Brown, C. J. (2020). Global trends in mangrove forest fragmentation. *Sci. Rep.* 10, 1–8. doi: 10.1038/s41598-020-63880-1

Buchner, B., Clark, A., Falconer, A., Macquarie, R., Meattle, C., and Wetherbee, C. (2019) *Global landscape of climate finance 2019.* Available at: http://resp.llas.ac.cn/C666/handle/2XK7JSWQ/242591 (Accessed July 2, 2022).

Bunting, P., Rosenqvist, A., Hilarides, L., Lucas, R. M., and Thomas, N. (2022). Global mangrove watch: Updated 2010 mangrove forest extent (v2.5). *Remote Sens.* 14, 1034. doi: 10.3390/rs14041034

Cadier, C., Bayraktarov, E., Piccolo, R., and Adame, M. F. (2020). Indicators of coastal wetlands restoration success: A systematic review. *Front. Mar. Sci.* 7. doi: 10.3389/fmars.2020.600220

Cadiz, P. L., Calumpong, H. P., Sinutok, S., and Chotikarn, P. (2020). Carbon storage potentials of natural and planted mangals in trang, Thailand. *Appl. Ecol. Environ. Res. h* 18, 4383–4403. doi: 10.15666/aeer/1803_43834403

Camacho, L. D., Gevaña, D. T., Sabino, L. L., Ruzol, C. D., Garcia, J. E., Camacho, A. C. D., et al. (2020). Sustainable mangrove rehabilitation: Lessons and insights from community-based management in the Philippines and Myanmar. *APN Sci. Bull.* 10, 18–25. doi: 10.30852/SB.2020.983

Cameron, C., Hutley, L. B., Friess, D. A., and Brown, B. (2019a). Community structure dynamics and carbon stock change of rehabilitated mangrove forests in sulawesi, Indonesia. *Ecol. Appl.* 29, 1–18. doi: 10.1002/eap.1810

Cameron, C., Hutley, L. B., Friess, D. A., and Munksgaard, N. C. (2019b). Hydroperiod, soil moisture and bioturbation are critical drivers of greenhouse gas fluxes and vary as a function of landuse change in mangroves of sulawesi, Indonesia. *Sci. Total Environ.* 654, 365–377. doi: 10.1016/j.scitotenv.2018.11.092

Caputo, A., and Kargina, M. (2021). A user-friendly method to merge scopus and web of science data during bibliometric analysis. *J. Mark. Anal.* 2021, 1–7. doi: 10.1057/S41270-021-00142-7

Castillo, J. A., MacKenzie, R., Manahan, J. R., and Castillo, J. (2022). Monitoring the sediment surface elevation change across a chronosequence of restored stands of tropical mangroves and their contemporary carbon sequestration in soil pool. *Forests*. 13, 241. doi: 10.3390/f13020241

CBD (Convention on Biological Diversity) (2010) *The strategic plan for biodiversity 2011-2020 and the aichi biodiversity targets COP 10 decision X/2.* Available at: https://www.cbd.int/decision/cop/?id=12268 (Accessed June 13, 2022).

Chou, M.-Q., Lin, W.-J., Lin, C.-W., Wu, H.-H., and Lin, H.-J. (2022). Allometric equations may underestimate the contribution of fine roots to mangrove carbon sequestration. *Sci. Total Environ.* 833, 155032. doi: 10.1016/j.scitotenv.2022.155032

Comer-Warner, S. A., Nguyen, A. T. Q., Nguyen, M. N., Wang, M., Turner, A., Le, H., et al. (2022). Restoration impacts on rates of denitrification and greenhouse gas fluxes from tropical coastal wetlands. *Sci. Total Environ.* 803, 149577. doi: 10.1016/j.scitotenv.2021.149577

Damastuti, E., de Groot, R., Debrot, A. O., and Silvius, M. J. (2022). Effectiveness of community-based mangrove management for biodiversity conservation: A case study from central Java, Indonesia. *Trees Forests People*. 7, 100202. doi: 10.1016/j.tfp.2022.100202

da Rosa, C. M., and Marques, M. C. M. (2022). How are biodiversity and carbon stock recovered during tropical forest restoration? Supporting the ecological paradigms and political context involved. *J. Nat. Conserv.* 65, 126115. doi: 10.1016/j.jnc.2021.126115

delos Santos, K. A., Avtar, R., Salmo, S., and Fujii, M. (2022). "Assessment of mangrove colonization of aquaculture ponds through satellite image analysis: Implications for mangrove management," in *Assessing, mapping and modelling mangrove ecosystem services in the Asia-pacific region*. Eds. R. Dasgupta (Singapore: Springer Nature). doi: 10.1007/978-981-19-2738-6_3

Díaz, S., Zafra-Calvo, N., Purvis, A., Verburg, P. H., Obura, D., Leadley, P., et al. (2020). Set ambitious goals for biodiversity and sustainability. *Sci. (1979)* 370, 411–413. doi: 10.1126/science.abe1530

Donato, D. C., Kauffman, J. B., Murdiyarso, D., Kurnianto, S., Stidham, M., and Kanninen, M. (2011). Mangroves among the most carbon-rich forests in the tropics. *Nat. Geosci.* 4, 293–297. doi: 10.1038/ngeo1123

Duncan, C., Primavera, J. H., Pettorelli, N., Thompson, J. R., Loma, R. J. A., and Koldewey, H. J. (2016). Rehabilitating mangrove ecosystem services: A case study on the relative benefits of abandoned pond reversion from panay island, Philippines. *Mar. pollut. Bull.* 109, 772–782. doi: 10.1016/j.marpolbul.2016.05.049

Egan, D., Hjerpe, E., and Abrams, J. (2011). *Human dimensions of ecological restoration: Integrating science, nature, and culture* (Washington D.C: Island Press).

Eger, A. M., Earp, H. S., Friedman, K., Gatt, Y., Hagger, V., Hancock, B., et al. (2022). The need, opportunities, and challenges for creating a standardized framework for marine restoration monitoring and reporting. *Biol. Conserv.* 266, 109429. doi: 10.1016/j.biocon.2021.109429

Ellison, A. M., Felson, A. J., and Friess, D. A. (2020). Mangrove rehabilitation and restoration as experimental adaptive management. *Front. Mar. Sci.* 7. doi: 10.3389/FMARS.2020.00327/BIBTEX

Estoque, R. C., Myint, S. W., Wang, C., Ishtiaque, A., Aung, T. T., Emerton, L., et al. (2018). Assessing environmental impacts and change in myanmar's mangrove ecosystem service value due to deforestation, (2000–2014). *Glob. Change Biol.* 24, 5391–5410. doi: 10.1111/gcb.14409

Friess, D. A., Gatt, Y. M., Ahmad, R., Brown, B. M., Sidik, F., and Wodehouse, D. (2022). Achieving ambitious mangrove restoration targets will need a transdisciplinary and evidence-informed approach. *One Earth* 5, 456–460. doi: 10.1016/j.oneear.2022.04.013

Friess, D. A., Richards, D. R., and Phang, V. X. H. (2016). Mangrove forests store high densities of carbon across the tropical urban landscape of Singapore. *Urban Ecosyst.* 19, 795–810. doi: 10.1007/s11252-015-0511-3

Friess, D. A., Yando, E. S., Alemu, J. B., Wong, L.-W., Soto, S. D., and Bhatia, N. (2020)Chapter 3 ecosystem services and disservices of mangrove forests and salt marshes. In: *Oceanography and marine biology* (Taylor & Francis). Available at: https://library.oapen.org/handle/20.500.12657/43146 (Accessed May 26, 2022).

Gandhi, S., and Jones, T. G. (2019). Identifying mangrove deforestation hotspots in south Asia, southeast Asia and Asia-pacific. *Remote Sens.* 11, 728. doi: 10.3390/RS11060728

Gann, G. D., McDonald, T., Walder, B., Aronson, J., Nelson, C. R., Jonson, J., et al. (2019). International principles and standards for the practice of ecological restoration. 2nd edition. *Restor. Ecol.* 27, S1–S46. doi: 10.1111/rec.13035

Garcia, K. B., Malabrigo, P. L., and Gevaña, D. T. (2014). "Philippines' mangrove ecosystem: status, threats and conservation," in *Mangrove ecosystems of Asia* (New York, NY: Springer New York), 81–94. doi: 10.1007/978-1-4614-8582-7_5

Gatt, Y. M., Andradi-Brown, D. A., Ahmadia, G. N., Martin, P. A., Sutherland, W. J., Spalding, M. D., et al. (2022). Quantifying the reporting, coverage and consistency of key indicators in mangrove restoration projects. *Front. For. Glob. Change* 5. doi: 10.3389/ffgc.2022.720394

Gevaña, D. T., Camacho, L. D., and Pulhin, J. M. (2018). Conserving mangroves for their blue carbon: Insights and prospects for community-based mangrove management in southeast Asia *Threats to Mangrove Forests, Coastal Research Library 25* C. Makowski and C. W. Frinkl (Cham, Switzerland: Springer Cham). 579–588. doi: 10.1007/978-3-319-73016-5_26

Giri, C., Zhu, Z., Tieszen, L. L., Singh, A., Gillette, S., and Kelmelis, J. A. (2008). Mangrove forest distributions and dynamics, (1975–2005) of the tsunami-affected region of Asia. *J. Biogeogr.* 35, 519–528. doi: 10.1111/j.1365-2699.2007.01806.x

Goessens, A., Satyanarayana, B., van der Stocken, T., Quispe Zuniga, M., Mohd-Lokman, H., Sulong, I., et al. (2014). Is matang mangrove forest in Malaysia sustainably rejuvenating after more than a century of conservation and harvesting management? *PloS One*. 9, e105069. doi: 10.1371/journal.pone.0105069

Gopalakrishnan, L., Satyanarayana, B., Chen, D., Wolswijk, G., Amir, A. A., Vandegehuchte, M., et al. (2021). Using historical archives and landsat imagery to explore changes in the mangrove cover of peninsular Malaysia between 1853 and 2018. *Remote Sens.* 13, 3403. doi: 10.3390/rs13173403

Guan, Y., Kang, R., and Liu, J. (2019). Evolution of the field of ecological restoration over the last three decades: A bibliometric analysis. *Restor. Ecol.* 27, 647–660. doi: 10.1111/REC.12899

Hai, N. T., Dell, B., Phuong, V. T., and Harper, R. J. (2020). Towards a more robust approach for the restoration of mangroves in Vietnam. *Ann. For. Sci.* 77, 1–18. doi: 10.1007/s13595-020-0921-0

Hamdan, O., Khali Aziz, H., and Mohd Hasmadi, I. (2014). L-band ALOS PALSAR for biomass estimation of matang mangroves, Malaysia. *Remote Sens. Enviro.* 155, 69–78. doi: 10.1016/j.rse.2014.04.029

Hashim, R., Kamali, B., Tamin, N. M., and Zakaria, R. (2010). An integrated approach to coastal rehabilitation: Mangrove restoration in sungai haji dorani, Malaysia. *Estuar. Coast. Shelf Sci.* 86, 118–124. doi: 10.1016/j.ecss.2009.10.021

Heberling, J. M., Miller, J. T., Noesgaard, D., Weingart, S. B., and Schigel, D. (2021). Data integration enables global biodiversity synthesis. *Proc. Natl. Acad. Sci.* 118, 1–7. doi: 10.1073/pnas.2018093118

Herr, D., Blum, J., Himes-Cornell, A., and Sutton-Grier, A. (2019). An analysis of the potential positive and negative livelihood impacts of coastal carbon offset projects. *J. Environ. Manage.* 235, 463–479. doi: 10.1016/j.jenvman.2019.01.067

Hochard, J. P., Hamilton, S., and Barbier, E. B. (2019). Mangroves shelter coastal economic activity from cyclones. *Proc. Natl. Acad. Sci. U.S.A.* 116, 12232–12237. doi: 10.1073/pnas.1820067116

Ho, Y. S., and Mukul, S. A. (2021). Publication performance and trends in mangrove forests: A bibliometric analysis. *Sustainability*. 13, 12532. doi: 10.3390/SU132212532

Hong, P. (2001). Reforestation of mangroves after severe impacts of herbicides during the Vietnam war: The case of can gio. *Unasylva*. 52 (207), 57–60. Available at: https://www.fao.org/3/y2795e/y2795e11.htm.

Howard, J., Sutton-Grier, A., Herr, D., Kleypas, J., Landis, E., Mcleod, E., et al. (2017). Clarifying the role of coastal and marine systems in climate mitigation. *Front. Ecol. Environ.* 15, 42–50. doi: 10.1002/fee.1451

Hughes, A. C. (2017). Understanding the drivers of southeast Asian biodiversity loss. *Ecosphere*. 8, e01624. doi: 10.1002/ecs2.1624

Ilman, M., Dargusch, P., and Dart, P. (2016). A historical analysis of the drivers of loss and degradation of indonesia's mangroves. *Land Use Policy*. 54, 448–459. doi: 10.1016/J.LANDUSEPOL.2016.03.010

Ilman, M., Wibisono, I. T. C., and Suryadiputra, I. N. N. (2011). *State of the art information on mangrove ecosystems in Indonesia* (Bogor, Indonesia: Wetlands International - Indonesia Programme). Available at: https://indonesia.wetlands.org/publications/state-of-the-art-information-on-mangrove-ecosystems-in-indonesia/

Iyandemye, J., and Thomas, M. P. (2019). Low income countries have the highest percentages of open access publication: A systematic computational analysis of the biomedical literature. *PloS One*. 14, e0220229. doi: 10.1371/journal.pone.0220229

Jordan, P., and Fröhle, P. (2022). Bridging the gap between coastal engineering and nature conservation? *J. Coast. Conserv*. 26, 4. doi: 10.1007/s11852-021-00848-x

Kamali, B., and Hashim, R. (2011). Mangrove restoration without planting. *Ecol. Eng.* 37, 387–391. doi: 10.1016/J.ECOLENG.2010.11.025

Kathiresan, K., and Rajendran, N. (2005). Coastal mangrove forests mitigated tsunami. *Estuar. Coast. Shelf Sci.* 65, 601–606. doi: 10.1016/j.ecss.2005.06.022

Katon, B. M., and Pomeroy, R. S. (2000). Rehabilitating the mangrove resources of cogtong bay, Philippines: A comanagement perspective. *Coast. Manage.* 28, 29–37. doi: 10.1080/089207500263620

Ken, S., Entani, T., Tsusaka, T. W., and Sasaki, N. (2020). Effect of REDD+ projects on local livelihood assets in keo seima and oddar meanchey, Cambodia. *Heliyon*. 6, e03802. doi: 10.1016/J.HELIYON.2020.E03802

Kodikara, K. A. S., Mukherjee, N., Jayatissa, L. P., Dahdouh-Guebas, F., and Koedam, N. (2017). Have mangrove restoration projects worked? An in-depth study in Sri Lanka. *Restor. Ecol.* 25, 705–716. doi: 10.1111/rec.12492

Kusumaningtyas, M. A., Kepel, T. L., Solihuddin, T., Lubis, A. A., Putra, A. D. P., Sugiharto, U., et al. (2022). Carbon sequestration potential in the rehabilitated mangroves in Indonesia. *Ecol. Res.* 37, 80–91. doi: 10.1111/1440-1703.12279

Lai, S., Loke, L. H. L., Hilton, M. J., Bouma, T. J., and Todd, P. A. (2015). The effects of urbanisation on coastal habitats and the potential for ecological engineering: A Singapore case study. *Ocean Coast. Manage.* 103, 78–85. doi: 10.1016/j.ocecoaman.2014.11.006

Lee, S. Y., Hamilton, S., Barbier, E. B., Primavera, J., and Lewis, R. R. (2019). Better restoration policies are needed to conserve mangrove ecosystems. *Nat. Ecol. Evol.* 3 (6), 870–872. doi: 10.1038/s41559-019-0861-y

Lewis, R. R. (2000). Ecologically based goal setting in mangrove forest and tidal marsh restoration. *Ecol. Eng.* 15, 191–198. doi: 10.1016/S0925-8574(00)00070-7

Lewis, R. R., Brown, B. M., and Flynn, L. L. (2019). "Methods and criteria for successful mangrove forest rehabilitation," in *Coastal wetlands* (Amsterdam, Netherlands: Elsevier), 863–887. doi: 10.1016/B978-0-444-63893-9.00024-1

Le Xuan, T., le Manh, H., Ba, H. T., Van, D., Duong Vu, H. T., Wright, D., et al. (2022). Wave energy dissipation through a hollow triangle breakwater on the coastal Mekong delta. *Ocean Eng.* 245, 110419. doi: 10.1016/j.oceaneng.2021.110419

Lin, R., Yu, S., and Hong, B. (2022). Socioeconomic patterns for global mangrove cover changes with multi-database analyses. *Wetlands* 42, 16. doi: 10.1007/s13157-022-01535-9

Long, J., Napton, D., Giri, C., and Graesser, J. (2014). A mapping and monitoring assessment of the philippines' mangrove forests from 1990 to 2010. *J. Coast. Res.* 30, 260–271. doi: 10.2112/JCOASTRES-D-13-00057.1

Lovelock, C. E., and Brown, B. M. (2019). Land tenure considerations are key to successful mangrove restoration. *Nat. Ecol. Evol.* 3, 1135. doi: 10.1038/s41559-019-0942-y

Luo, J., Sun, Z., Lu, L., Xiong, Z., Cui, L., and Mao, Z. (2022). Rapid expansion of coastal aquaculture ponds in southeast Asia: Patterns, drivers and impacts. *J. Environ. Manage.* 315, 115100. doi: 10.1016/j.jenvman.2022.115100

Macintosh, D. J., Ashton, E. C., and Havanon, S. (2002). Mangrove rehabilitation and intertidal biodiversity: a study in the ranong mangrove ecosystem, Thailand. *Estuar. Coast. Shelf Sci.* 55, 331–345. doi: 10.1006/ecss.2001.0896

Macreadie, P. I., Robertson, A. I., Spinks, B., Adams, M. P., Atchison, J. M., Bell-James, J., et al. (2022). Operationalizing marketable blue carbon. *One Earth*. 5, 485–492. doi: 10.1016/j.oneear.2022.04.005

Maina, J. M., Bosire, J. O., Kairo, J. G., Bandeira, S. O., Mangora, M. M., Macamo, C., et al. (2021). Identifying global and local drivers of change in mangrove cover and the implications for management. *Glob. Ecol. Biogeogr.* 30, 2057–2069. doi: 10.1111/geb.13368

Martin, F., Dutrieux, É., and Debry, A. (1990). Natural recolonization of a chronically oil-polluted mangrove soil after a de-pollution process. *Ocean Shoreline Manage.* 14, 173–190. doi: 10.1016/0951-8312(90)90033-E

Matheka, D., Nderitu, J., Mutonga, D., Otiti, M., Siegel, K., and Demaio, A. (2014). Open access: academic publishing and its implications for knowledge equity in Kenya. *Glob. Health*. 10, 26. doi: 10.1186/1744-8603-10-26

Matsui, N., Suekuni, J., Havanond, S., Nishimiya, A., Yanai, J., and Kosaki, T. (2008). Determination of soil-related factors controlling initial mangrove (*Rhizophora apiculata* BL.) growth in an abandoned shrimp pond. *Soil Sci. Plant Nutr.* 54, 301–309. doi: 10.1111/j.1747-0765.2007.00238.x

Matsui, N., Suekuni, J., Nogami, M., Havanond, S., and Salikul, P. (2010). Mangrove rehabilitation dynamics and soil organic carbon changes as a result of full hydraulic restoration and re-grading of a previously intensively managed shrimp pond. *Wetl. Ecol. Manage.* 18, 233–242. doi: 10.1007/s11273-009-9162-6

Mohd Razali, S., Radzi, M. A., Marin, A., and Samdin, Z. (2021). A bibliometric analysis of tropical mangrove forest land use change from 2010 to 2020. *Environ. Dev. Sustain*. doi: 10.1007/s10668-021-01935-7

Moher, D., Liberati, A., Tetzlaff, J., Altman, D. G., and Group, T. P. (2009). Preferred reporting items for systematic reviews and meta-analyses: The PRISMA statement. *PloS Med.* 6, e1000097. doi: 10.1371/JOURNAL.PMED.1000097

Mursyid, H., Daulay, M. H., Pratama, A. A., Laraswati, D., Novita, N., Malik, A., et al. (2021). Governance issues related to the management and conservation of mangrove ecosystems to support climate change mitigation actions in Indonesia. *For. Policy Econ.* 133, 1–6. doi: 10.1016/j.forpol.2021.102622

Nam, V. N., Sasmito, S. D., Murdiyarso, D., Purbopuspito, J., and MacKenzie, R. A. (2016). Carbon stocks in artificially and naturally regenerated mangrove ecosystems in the Mekong delta. *Wetl. Ecol. Manage.* 24, 231–244. doi: 10.1007/S11273-015-9479-2

Nawari, ., Syahza, A., and Siregar, Y. I. (2021). Community-based mangrove forest management as ecosystem services provider for reducing CO_2 emissions with carbon credit system in bengkalis district, riau, Indonesia. *J. Phys. Conf. Ser.* 2049, 12074. doi: 10.1088/1742-6596/2049/1/012074

Nguyen, T. P. (2018). Melaleuca entrapping microsites as a nature-based solution to coastal erosion: A pilot study in kien giang, Vietnam. *Ocean Coast. Manage.* 155, 98–103. doi: 10.1016/j.ocecoaman.2018.02.005

Nguyen, H. Y. T., Cao, D. M., and Schmitt, K. (2013). Soil particle-size composition and coastal erosion and accretion study in soc trang mangrove forests. *J. Coast. Conserv.* 17, 93–104. doi: 10.1007/s11852-012-0221-4

Noor, N. M., and Abdul Maulud, K. N. (2022). Coastal vulnerability: A brief review on integrated assessment in southeast Asia. *J. Mar. Sci. Eng.* 10, 595. doi: 10.3390/jmse10050595

Oka, S., Doi, H., Miyamoto, K., Hanahara, N., Sado, T., and Miya, M. (2021). Environmental DNA metabarcoding for biodiversity monitoring of a highly diverse tropical fish community in a coral reef lagoon: Estimation of species richness and detection of habitat segregation. *Environ. DNA*. 3, 55–69. doi: 10.1002/edn3.132

Ong, S. (2021). Southeast Asian countries join forces for scientific strength. *Nature*. 591, S30–S32. doi: 10.1038/d41586-021-00670-3

Orchard, S. E., Stringer, L. C., and Quinn, C. H. (2016). Mangrove system dynamics in southeast Asia: Linking livelihoods and ecosystem services in Vietnam. *Reg. Environ. Change*. 16, 865–879. doi: 10.1007/s10113-015-0802-5

Palis, H., Pasicolan, S., and Villamor, C. (2014). *Proceedings of the 1st ASEAN congress on mangrove research and development, 3-7 December 2012, Manila, Philippines*. Eds. H. G. Palis, S. A. Pasicolan and C. I. Villamor (Metro Manila, Philippines: Department of Environment and Natural Resources - Biodiversity Management Bureau and Conservation International Philippines). Available at: https://erdb.denr.gov.ph/wp-content/uploads/2015/06/asean_mangrove_proceedings.pdf

Piwowar, H., Priem, J., Larivière, V., Alperin, J. P., Matthias, L., Norlander, B., et al. (2018). The state of OA: A large-scale analysis of the prevalence and impact of open access articles. *PeerJ*. 6, e4375. doi: 10.7717/peerj.4375

Polanco Fernández, A., Marques, V., Fopp, F., Juhel, J., Borrero-Pérez, G. H., Cheutin, M., et al. (2021). Comparing environmental DNA metabarcoding and underwater visual census to monitor tropical reef fishes. *Environ. DNA*. 3, 142–156. doi: 10.1002/edn3.140

Polidoro, B. A., Carpenter, K. E., Collins, L., Duke, N. C., Ellison, A. M., Ellison, J. C., et al. (2010). The loss of species: Mangrove extinction risk and geographic areas of global concern. *PloS One*. 5, e10095. doi: 10.1371/JOURNAL.PONE.0010095

Powell, M. A. (2021). Singapore's lost coast: Land reclamation, national development and the erasure of human and ecological communities 1822-present. *Environ. Hist.* 27, 635–663. doi: 10.3197/096734019X15631846928710

Primavera, J. H. (1995). Mangroves and brackishwater pond culture in the Philippines. *Hydrobiologia*. 295, 303–309. doi: 10.1007/BF00029137

Primavera, J. H. (2000). Development and conservation of Philippine mangroves: institutional issues. *Ecol. Econ.* 35, 91–106. doi: 10.1016/S0921-8009(00)00170-1

Primavera, J. H., dela Cruz, M., Montilijao, C., Consunji, H., dela Paz, M., Rollon, R. N., et al. (2016). Preliminary assessment of post-haiyan mangrove damage and short-term recovery in Eastern samar, central Philippines. *Mar. pollut. Bull.* 109, 744–750. doi: 10.1016/J.MARPOLBUL.2016.05.050

Primavera, J. H., and Esteban, J. M. A. (2008). A review of mangrove rehabilitation in the Philippines: Successes, failures and future prospects. *Wetlands Ecol. Manage.* 16, 345–358. doi: 10.1007/s11273-008-9101-y

Primavera, J. H., Friess, D. A., van Lavieren, H., and Lee, S. Y. (2019). "The mangrove ecosystem," in *World seas: An environmental evaluation* (London, UK: Elsevier), 1–34. doi: 10.1016/B978-0-12-805052-1.00001-2

Primavera, J., Rollon, R., and Samson, M. (2011). "The pressing challenges of mangrove rehabilitation: pond reversion and coastal protection," in *Treatise on estuarine and coastal science*. Eds. E. Wolanski and D. McLusky (Waltham, MA, USA: Waltham: Academic Press), 217–244.

Primavera, J. H., Savaris, J. P., Bajoyo, B. E., Coching, J. D., Curnick, D. J., Golbeque, R. L., et al. (2012). *Manual on community-based mangrove rehabilitation*

– *mangrove manual series no. 1* (Iloilo, Philippines: Zoological Society of London-Philippines).

Quevedo, J. M. D., Uchiyama, Y., and Kohsaka, R. (2020). Perceptions of local communities on mangrove forests, their services and management: Implications for eco-DRR and blue carbon management for Eastern samar, Philippines. *J. For. Res.* 25, 1–11. doi: 10.1080/13416979.2019.1696441

Quevedo, J. M. D., Uchiyama, Y., and Kohsaka, R. (2021a). A blue carbon ecosystems qualitative assessment applying the DPSIR framework: Local perspective of global benefits and contributions. *Mar. Policy*. 128, 104462. doi: 10.1016/j.marpol.2021.104462

Quevedo, J. M. D., Uchiyama, Y., and Kohsaka, R. (2021b). Local perceptions of blue carbon ecosystem infrastructures in panay island, Philippines. *Coast. Eng.* 63, 227–247. doi: 10.1080/21664250.2021.1888558

Reddy, C. S. (2021). Remote sensing of biodiversity: What to measure and monitor from space to species? *Biodivers. Conserv.* 30, 2617–2631. doi: 10.1007/s10531-021-02216-5

Richards, D. R., and Friess, D. A. (2016). Rates and drivers of mangrove deforestation in southeast Asia 2000-2012. *Proc. Natl. Acad. Sci. U.S.A.* 113, 344–349. doi: 10.1073/PNAS.1510272113/-/DCSUPPLEMENTAL

Ridlo, I. A., Kurniawan, N., and Retnaningdyah, C. (2020). Communities structure of fish in some mangrove ecosystem as a result of the restoration in southern beach of malang, East Java Indonesia. *J. Phys. Conf. Ser.* 1665, 12020. doi: 10.1088/1742-6596/1665/1/012020

Rittibhonbhun, N., Chansanoh, P., Tongrak, S., and Suwannatchote, R. (1993). *Community-based mangrove rehabilitation and management: A case study in sikao district, trang province, southern Thailand* (Regional Development Dialogue) 14, 111–124.

Romañach, S. S., DeAngelis, D. L., Koh, H. L., Li, Y., Teh, S. Y., Raja Barizan, R. S., et al. (2018). Conservation and restoration of mangroves: Global status, perspectives, and prognosis. *Ocean Coast. Manage.* 154, 72–82. doi: 10.1016/j.ocecoaman.2018.01.009

Salem, M. E., and Mercer, D. E. (2012). The economic value of mangroves: a meta-analysis. *Sustainability*. 4, 359–383. doi: 10.3390/su4030359

Salmo, S. G. (2019) *Mangrove blue carbon in the Verde island passage. nature is 30 (Climate strategy)* (Conservation International Philippines). Available at: https://archium.ateneo.edu/es-faculty-pubs (Accessed February 7, 2022).

Salmo, S. G. (2021). "Assessment of typhoon impacts and post-typhoon recovery in Philippine mangroves: lessons and challenges for adaptive management," in *Dynamic sedimentary environments of mangrove coasts* (Amsterdam, Netherlands: Elsevier), 539–562. doi: 10.1016/B978-0-12-816437-2.00022-7

Salmo, S. G., Gevaña, G. D. T., Halun, S. Z. B., Castillo, J. A. A., Nuñez, E. A.Jr., Pangilinan, M. J. M., et al. (2021). *Revisiting learnings and envisioning Philippine mangroves in 2030: Proceedings of the 1st national state of the mangrove summit* (Metro Manila, Philippines: Department of Environment and Natural Resources - Biodiversity Management Bureau and Conservation International Philippines).

Salmo, S. G., and Gianan, E. L. D. (2019). Post-disturbance carbon stocks and rates of sequestration: Implications on "blue carbon" estimates in Philippine mangroves. *Philippine Sci. Lett.* 12, 122–132. Available at: https://www.philsciletters.net/2019/PSL%202019-vol12-no02-p122-132-Salmo%20and%20Gianan.pdf.

Salmo, S. G., Lovelock, C., and Duke, N. C. (2013). Vegetation and soil characteristics as indicators of restoration trajectories in restored mangroves. *Hydrobiologia*. 720, 1–18. doi: 10.1007/s10750-013-1617-3

Salmo, S. G., Tibbetts, I., and Duke, N. C. (2017). Colonization and shift of mollusc assemblages as a restoration indicator in planted mangroves in the Philippines. *Biodivers. Conserv.* 26, 865–881. doi: 10.1007/s10531-016-1276-6

Salmo, S. G., Tibbetts, I. R., and Duke, N. C. (2018). Nekton communities as indicators of habitat functionality in Philippine mangrove plantations. *Mar. Freshw. Res.* 69, 477–485. doi: 10.1071/MF17116

Sannigrahi, S., Zhang, Q., Pilla, F., Joshi, P. K., Basu, B., Keesstra, S., et al. (2020). Responses of ecosystem services to natural and anthropogenic forcings: A spatial regression based assessment in the world's largest mangrove ecosystem. *Sci. Total Environ.* 715, 137004. doi: 10.1016/j.scitotenv.2020.137004

Sartimbul, A., Ningtias, S. W., Dewi, C. S. U., Rahman, M. A. A., Yona, D., Sari, S. H. J., et al. (2021). Monitoring of sedimentation on geosynthetic bags installation area in banyuurip mangrove center, ujung pangkah, gresik, Indonesia. *ILMU KELAUTAN: Indonesian J. Mar. Sci.* 26, 173–181. doi: 10.14710/ik.ijms.26.3.173-181

Sasmito, S. D., Sillanpää, M., Hayes, M. A., Bachri, S., Saragi-Sasmito, M. F., Sidik, F., et al. (2020). Mangrove blue carbon stocks and dynamics are controlled by hydrogeomorphic settings and land-use change. *Glob. Change Biol.* 26, 3028–3039. doi: 10.1111/gcb.15056

Schmitt, K., Albers, T., Pham, T. T., and Dinh, S. C. (2013). Site-specific and integrated adaptation to climate change in the coastal mangrove zone of soc trang province, Viet nam. *J. Coast. Conserv.* 17, 545–558. doi: 10.1007/s11852-013-0253-4

Sharma, S., MacKenzie, R. A., Tieng, T., Soben, K., Tulyasuwan, N., Resanond, A., et al. (2020). The impacts of degradation, deforestation and restoration on mangrove ecosystem carbon stocks across Cambodia. *Sci. Total Environ.* 706, 135416. doi: 10.1016/j.scitotenv.2019.135416

Sharma, S., Nadaoka, K., Nakaoka, M., Uy, W. H., MacKenzie, R. A., Friess, D. A., et al. (2017). Growth performance and structure of a mangrove afforestation project on a former seagrass bed, Mindanao island, Philippines. *Hydrobiologia*. 803, 359–371. doi: 10.1007/s10750-017-3252-x

Sidik, F., Fernanda Adame, M., and Lovelock, C. E. (2019). Carbon sequestration and fluxes of restored mangroves in abandoned aquaculture ponds. *J. Indian Ocean Reg.* 15, 177–192. doi: 10.1080/19480881.2019.1605659

Sillanpää, M., Vantellingen, J., and Friess, D. A. (2017). Vegetation regeneration in a sustainably harvested mangrove forest in West Papua, Indonesia. *For. Ecol. Manage.* 390, 137–146. doi: 10.1016/j.foreco.2017.01.022

10.3389/fmars.2022.95147110.3389/fmars.2022.921542 w/CCSlamet, N. S., Dargusch, P., Aziz, A. A., and Wadley, D. (2020). Mangrove vulnerability and potential carbon stock loss from land reclamation in Jakarta bay, Indonesia. *Ocean Coast. Manage.* 195, 105283. doi: 10.1016/j.ocecoaman.2020.105283

Sodhi, N. S., Posa, M. R. C., Lee, T. M., Bickford, D., Koh, L. P., and Brook, B. W. (2010). The state and conservation of southeast Asian biodiversity. *Biodivers. Conserv.* 19, 317–328. doi: 10.1007/s10531-009-9607-5

Song, A. M., Dressler, W. H., Satizábal, P., and Fabinyi, M. (2021). From conversion to conservation to carbon: The changing policy discourse on mangrove governance and use in the Philippines. *J. Rural Stud.* 82, 184–195. doi: 10.1016/j.jrurstud.2021.01.008

Spalding, M., Kainuma, M., and Collins, L. (2010). *World atlas of mangroves* (London: Earthscan). doi: 10.4324/9781849776608

Spalding, M. D., and Leal, M. (2021) *The state of the world's mangroves* (Global Mangrove Alliance). Available at: https://www.mangrovealliance.org/mangrove-forests/ (Accessed November 8, 2021).

Spalding, M., and Parrett, C. L. (2019). Global patterns in mangrove recreation and tourism. *Mar. Policy*. 110, 103540. doi: 10.1016/j.marpol.2019.103540

Suman, D. O. (2019). "Mangrove management," in *Coastal wetlands* (Amsterdam, Netherlands: Elsevier), 1055–1079. doi: 10.1016/B978-0-444-63893-9.00031-9

Suripin, S., Sugianto, D., and Helmi, M. (2017). Coastal restoration with environmentally friendly permeable breakwater. *Advanced Sci. Lett.* 23, 2323–2325. doi: 10.1166/asl.2017.8733

Su, J., Yin, B., Chen, L., and Gasparatos, A. (2022). Priority areas for mixed-species mangrove restoration: the suitable species in the right sites. *Environ. Res. Lett.* 17, 065001. doi: 10.1088/1748-9326/ac6b48

Taddeo, S., and Dronova, I. (2018). Indicators of vegetation development in restored wetlands. *Ecol. Indic.* 94, 454–467. doi: 10.1016/j.ecolind.2018.07.010

Tanaka, N., Sasaki, Y., Mowjood, M. I. M., Jinadasa, K. B. S. N., and Homchuen, S. (2007). Coastal vegetation structures and their functions in tsunami protection: Experience of the recent Indian ocean tsunami. *Landsc. Ecol. Eng.* 3, 33–45. doi: 10.1007/s11355-006-0013-9

Tan, Y.-L., Chen, J.-E., Yiew, T.-H., and Habibullah, M. S. (2022). Habitat change and biodiversity loss in south and southeast Asian countries. *Environ. Sci. pollut. Res.* doi: 10.1007/s11356-022-20054-y

Teutli-Hernández, C., Herrera-Silveira, J. A., Cisneros-de la Cruz, D. J., and Román-Cuesta, R. (2020). *Mangrove ecological restoration guide: Lessons learned. mainstreaming wetlands into the climate agenda: A multilevel approach (SWAMP).* (Bogor, Indonesia: CIFOR/CINVESTAV-IPN/UNAM-Sisal/PMC)42p.

Thakur, S., Mondal, I., Bar, S., Nandi, S., Ghosh, P. B., Das, P., et al. (2021). Shoreline changes and its impact on the mangrove ecosystems of some islands of Indian sundarbans, north-East coast of India. *J. Clean. Prod.* 284, 124764. doi: 10.1016/j.jclepro.2020.124764

Thampanya, U., Vermaat, J. E., Sinsakul, S., and Panapitukkul, N. (2006). Coastal erosion and mangrove progradation of southern Thailand. *Estuar. Coast. Shelf Sci.* 68, 75–85. doi: 10.1016/j.ecss.2006.01.011

Then, A. Y.-H., Adame, M. F., Fry, B., Chong, V. C., Riekenberg, P. M., Mohammad Zakaria, R., et al. (2021). Stable isotopes clearly track mangrove inputs and food web changes along a reforestation gradient. *Ecosystems*. 24, 939–954. doi: 10.1007/s10021-020-00561-0

Thieu Quang, T., and Mai Trong, L. (2020). Monsoon wave transmission at bamboo fences protecting mangroves in the lower Mekong delta. *Appl. Ocean Res.* 101, 102259. doi: 10.1016/j.apor.2020.102259

Thompson, B. S. (2018). The political ecology of mangrove forest restoration in Thailand: Institutional arrangements and power dynamics. *Land Use Policy*. 78, 503–514. doi: 10.1016/j.landusepol.2018.07.016

Thompson, B. S., Primavera, J. H., and Friess, D. A. (2017). Governance and implementation challenges for mangrove forest payments for ecosystem services (PES): Empirical evidence from the Philippines. *Ecosyst. Serv.* 23, 146–155. doi: 10.1016/J.ECOSER.2016.12.007

Tinh, P. H., MacKenzie, R. A., Hung, T. D., Hanh, N. T. H., Hanh, N. H., Manh, D. Q., et al. (2022). Distribution and drivers of Vietnam mangrove deforestation from 1995 to 2019 *Mitig. Adapt. Strateg. Glob. Change* 27, 29. doi: 10.1007/s11027-022-10005-w

Ulfa, M., Ikejima, K., Poedjirahajoe, E., Faida, L. R. W., and Harahap, M. M. (2018). Effects of mangrove rehabilitation on density of Scylla spp. (mud crabs) in Kuala langsa, aceh, Indonesia. *Reg. Stud. Mar. Sci.* 24, 296–302. doi: 10.1016/j.rsma.2018.09.005

Valenzuela, R. B., Yeo-Chang, Y., Park, M. S., and Chun, J.-N. (2020). Local people's participation in mangrove restoration projects and impacts on social capital and livelihood: a case study in the Philippines. *Forests* 11, 580. doi: 10.3390/f11050580

Valiela, I., Bowen, J. L., and York, J. K. (2001). Mangrove forests: One of the world's threatened major tropical environments: At least 35% of the area of mangrove forests has been lost in the past two decades, losses that exceed those for tropical rain forests and coral reefs, two other well-known threatened environments. *BioScience*. 51, 807–815. doi: 10.1641/0006-3568(2001)051[0807:MFOOTW]2.0.CO;2

Villamayor, B. M. R., Rollon, R. N., Samson, M. S., Albano, G. M. G., and Primavera, J. H. (2016). Impact of haiyan on Philippine mangroves: Implications to the fate of the widespread monospecific rhizophora plantations against strong typhoons. *Ocean Coast. Manag.* 132, 1–14. doi: 10.1016/j.ocecoaman.2016.07.011

Walters, B. B. (2003). People and mangroves in the Philippines: fifty years of coastal environmental change. *Environ. Conserv.* 30, 293–303. doi: 10.1017/S0376892903000298

Waltham, N. J., Elliott, M., Lee, S. Y., Lovelock, C., Duarte, C. M., Buelow, C., et al. (2020). UN Decade on ecosystem restoration 2021–2030—What chance for success in restoring coastal ecosystems? *Front. Mar. Sci.* 7. doi: 10.3389/FMARS.2020.00071/BIBTEX

Wee, A. K. S., Mori, G. M., Lira, C. F., Núñez-Farfán, J., Takayama, K., Faulks, L., et al. (2019). The integration and application of genomic information in mangrove conservation. *Conserv. Biol.* 33, 206–209. doi: 10.1111/cobi.13140

Wernick, I. K., and Kauppi, P. E. (2022). Storing carbon or growing forests? *Land Use Policy*. 121, 106319. doi: 10.1016/j.landusepol.2022.106319

Winterwerp, J. C., Albers, T., Anthony, E. J., Friess, D. A., Mancheño, A. G., Moseley, K., et al. (2020). Managing erosion of mangrove-mud coasts with permeable dams – lessons learned. *Ecol. Eng.* 158, 106078. doi: 10.1016/j.ecoleng.2020.106078

Winterwerp, J. C., Erftemeijer, P. L. A., Suryadiputra, N., van Eijk, P., and Zhang, L. (2013). Defining eco-morphodynamic requirements for rehabilitating eroding mangrove-mud coasts. *Wetlands*. 33, 515–526. doi: 10.1007/s13157-013-0409-x

Wodehouse, D. C. J., and Rayment, M. B. (2019). Mangrove area and propagule number planting targets produce sub-optimal rehabilitation and afforestation outcomes. *Estuar. Coast. Shelf Sci.* 222, 91–102. doi: 10.1016/j.ecss.2019.04.003

Worthington, T., and Spalding, M. (2018). *Mangrove restoration potential: A global map highlighting a critical opportunity*. (Virginia, USA: IUCN/The Nature Conservancy/University of Cambridge). doi: 10.17863/CAM.39153

Zari, M. P., Kiddle, G. L., Blaschke, P., Gawler, S., and Loubser, D. (2019). Utilising nature-based solutions to increase resilience in pacific ocean cities. *Ecosyst. Serv.* 38, 100968. doi: 10.1016/j.ecoser.2019.100968

Zhang, Y. S., Cioffi, W. R., Cope, R., Daleo, P., Heywood, E., Hoyt, C., et al. (2018). A global synthesis reveals gaps in coastal habitat restoration research. *Sustainability (Switzerland)* 10, 1–15. doi: 10.3390/su10041040

frontiers | Frontiers in Marine Science

TYPE Original Research
PUBLISHED 25 November 2022
DOI 10.3389/fmars.2022.1033200

Check for updates

OPEN ACCESS

EDITED BY
Richard MacKenzie,
Forest Service (USDA), United States

REVIEWED BY
Drandreb Earl O. Juanico,
Technological Institute of the Philippines, Philippines
Dixon Gevaña,
University of the Philippines Los Baños, Philippines

*CORRESPONDENCE
Amy Yee-Hui Then
amy_then@um.edu.my
Melanie C. Austen
melanie.austen@plymouth.ac.uk

SPECIALTY SECTION
This article was submitted to Marine Conservation and Sustainability, a section of the journal Frontiers in Marine Science

RECEIVED 31 August 2022
ACCEPTED 11 November 2022
PUBLISHED 25 November 2022

CITATION
Lee SL, Then AY-H, Goh HC, Hattam C, Edwards-Jones A and Austen MC (2022) Strengthened multi-stakeholder linkages in valuation studies is critical for improved decision making outcomes for valuable mangroves – The Malaysian case study.
Front. Mar. Sci. 9:1033200.
doi: 10.3389/fmars.2022.1033200

Strengthened multi-stakeholder linkages in valuation studies is critical for improved decision making outcomes for valuable mangroves – The Malaysian case study

Soon Loong Lee[1], Amy Yee-Hui Then[1*], Hong Ching Goh[2], Caroline Hattam[3], Andrew Edwards-Jones[4] and Melanie C. Austen[5*]

[1]Institute of Biological Sciences, Faculty of Science, Universiti Malaya, Kuala Lumpur, Malaysia, [2]Department of Urban & Regional Planning, Faculty of Built Environment, Universiti Malaya, Kuala Lumpur, Malaysia, [3]ICF, Plymouth, United Kingdom, [4]Plymouth Marine Laboratory, Plymouth, United Kingdom, [5]Plymouth School of Biological and Marine Sciences, Faculty of Science and Engineering, University of Plymouth, Plymouth, United Kingdom

Mangrove forests in Southeast Asia are continuously declining as a result of unsustainable practices, partly due to limited recognition of the value of mangrove services in land use decision making. Valuation practitioners have assumed that monetary valuation should inform local and national decision makers to ensure sustainable management of mangrove resources. For ecosystem service valuation to be of use to decision makers, best practices should be adhered to such as having straightforward policy questions and strong stakeholder engagement from the onset of valuation studies, suitable choice of valuation methodologies, and the ability to effectively demonstrate causal links between drivers of ecosystem health, change, and resource users. This study, focusing on the Malaysian case study, assessed the effectiveness and challenges of local ecosystem service valuation studies in informing mangrove management decisions against a set of global best practices. A systematic review approach was undertaken to identify relevant Malaysian mangrove ecosystem service valuation studies. Of 184 studies identified, only 17 provided monetary values for mangrove ecosystem services. These studies valued nine different mangrove ecosystem services, with the cultural ecosystem services of tourism being the most frequently valued. Most of the valuation studies were designed to raise awareness of the value of ecosystems (64.7%). Other intended uses included determining appropriate charging rates for mangrove uses (17.6%), comparing the costs and benefits of different environmental uses (11.8%), and providing a justification and support for certain decision making (5.9%). Overall, mangrove valuation studies in Malaysia were characterized by weak multi-stakeholder engagement, nonstandardized valuation units across the whole country, limited dissemination of the valuation outcome, and cursory references to the potential use of mangrove ecosystem services. Most of the studies did not exert apparent influence on mangrove management. Future valuation studies in Malaysia and the Southeast Asian region should aim to build more robust engagement between valuation practitioners and key stakeholder groups, especially decision makers, at all stages of the study process and incorporate a clear dissemination strategy for sharing results.

KEYWORDS

ecosystem service assessment, policy making, southeast Asia, natural resources, decision making, result dissemination

1 Introduction

Worldwide, mangrove ecosystems are in decline due largely to unsustainable anthropogenic activities and the effects of climate change (Gilman et al., 2008; Friess et al., 2019). One of the factors contributing to the continual loss of mangrove ecosystems is the limited understanding of the value of mangrove ecosystem services, and their consequent omission in public decision and policy making (Brander et al., 2012). This is despite mangroves being widely recognized as a vital nature-based solution to mitigate climate change impacts, particularly for their ability to sequester and store blue carbon (McLeod and Salm, 2006; Zeng et al., 2021). Decision-makers worldwide have thus been urged to increase efforts to conserve remaining mangrove forests and rehabilitate degraded ones (Duarte et al., 2020; Ellison et al., 2020).

Ecosystem services are the direct and indirect contributions of ecosystems to human well-being (TEEB, 2010). Notable ecosystem services from mangroves include provisioning services such as timber extraction and coastal fishery production; regulatory services such as storm surge and erosion protection, and climate regulation; supporting services such as carbon sequestration and primary production; and cultural services such as recreation, and knowledge-based activities. The practice of ecosystem service valuation quantifies the flows of goods and services from natural capital assets (including mangroves) and assumes that they are manageable by stakeholders and decision makers (Daily et al., 2009; Tisdell and Xue, 2013). In doing so, valuation aims to ensure that the value of ecosystems and the services they provide is better recognised in policy decision-making processes (Daily et al., 2009; Pendleton et al., 2015). For example, valuation of ecosystem services can support decision-makers to make comparisons between alternative management regimes (van Oudenhoven et al., 2015). Valuation has also enabled cost estimation for the purposes of setting insurance policies and assessing the cost of climate disaster prevention (Bayraktarov et al., 2016; Beck et al., 2020). In the context of mangroves, ecosystem service valuation studies appear to have gained higher traction in recent years to support decision making (Barbier et al., 2011; Mukherjee et al., 2014; Himes-Cornell et al., 2018a).

Common methods used for natural resource valuation can be categorized into two broad groups: revealed preference methods (such as market price, travel cost and production function) and stated preference methods (such as contingent valuation and conjoint analysis). The former rely on individual preferences for services with definite market value, whereas the latter survey individuals' stated preferences in value for a given change in a natural resource or services (DEFRA, 2007). In the context of mangroves, the benefit transfer method appears to be one of the most commonly used valuation methods (Himes-Cornell et al., 2018b). Benefit transfer allows researchers to transfer ecosystem service values calculated in previous studies for ecosystems similar to the one(s) they are studying. The method may circumvent the need for costly and time-intensive field valuation studies (TEEB, 2010). However, benefit transfer has a number of shortcomings. For example, values may be inflated as they are estimated from global values, such as those from Costanza et al. (2014), who originally created values by statistically extrapolating value estimates to entire biomes (Pendleton et al., 2015). Moreover, benefit transfer values can be laden with inaccuracies due to the use of values for one site that were originally calculated for another biophysically, ecologically and socioeconomically distinct location (Emerton, 2014; Himes-Cowell et al., 2018b). Valuation studies that rely heavily on benefit transfer data (secondary data) also suffer from insufficient primary studies or meta-analyses that include comprehensive socio-economic information (Himes-Cornell et al., 2018b), which could be vital to decision making.

To date, a number of ecosystem service valuation guidelines are available that are intended to ensure that the true value of ecosystems services provided are properly taken into account in supporting decision making (e.g. DEFRA, 2007; Stelk and Christie, 2014; Schuster and Doerr, 2015). Several enabling conditions and lessons learnt for ecosystem service valuations have been identified to ensure such studies are effective or appropriate to the relevant decision makers (Laurans et al., 2013; Waite et al., 2015; Torres and Hanley, 2017). One key recommendation for valuation practitioners is to craft a sound valuation methodology that is suitable to the local context and can effectively convey relevant information to decision makers. Having clear policy questions from the onset of valuation studies will improve relevance of results or recommendations and facilitate their use (McVittie and Moran, 2010; Waite et al., 2015). Policy questions can address, for example, the ecosystem services at stake, the policy options for these services, or the effects of policy change on them (Schuster and Doerr, 2015; Waite et al., 2015). Strong stakeholder engagement and local partnerships (Torres and Hanley, 2017), and clear presentation of methods and limitations (Lange and Jiddawi, 2009; Himes-Cornell et al., 2018a) are important characteristics for increased uptake of ecosystem service valuation studies. This highlights the importance of transdisciplinary cooperation, and the need to combine knowledge and data from different sources and multiple stakeholders, such as from economists, political, communication and natural scientists.

Valuation practitioners may have limited understanding of the circumstances and realities of policy making, the political climate, concerns around rights and the needs of stakeholders and thus unintentionally create barriers to effective use of ecosystem valuation outputs (Kenter et al., 2015; Torres and Hanley, 2017). Many types of information are required to support land-use decision making such as budgets, details of social, political and equity concerns, and understanding of how decisions result in benefits to the beneficiaries and wider stakeholders, often in a constrained time period (Rogers et al., 2015). Decision makers have often lamented that results from valuation studies are not sufficiently relevant to inform socially optimal decisions (Vatn and Bromley, 1994; Torres and Hanley, 2017). The lack of uptake of valuation outputs can be further exacerbated by decision makers' lack of familiarity in the language and axioms of ecosystem service valuation (Laurans et al., 2013). Incorporating causal chains in an ecosystem service assessment has been advocated as a means to help decision makers by expanding the focus beyond ecological outcomes to social outcomes caused by the ecological changes (Wainger and Mazzotta, 2011; Olander et al., 2015).

While a number of enabling conditions can be facilitated by valuation practitioners, there are external conditions that are beyond their control, such as the local political climate, governance, and economic dependence on the ecosystem services (Waite et al., 2015). Good governance within and among governments and other stakeholder organizations is needed to facilitate the use of scientific information in decision making (Nursey-Bray et al., 2014). The existence of legal authorities that develop conservation-oriented policy and legislation can further levy the incentive to use valuation results in the form of establishing protected areas or charging entrance fees (Waite et al., 2015). For example, as a result of a valuation study in close consultation with local communities, the federal government of Mexico had created marine protected areas near Cancun and approved the setting up of marine park entry fees to finance park infrastructure, staff, and environmental education campaigns (Rivera-Planter and Muñoz-Piña, 2005). On the other hand, in instances where government capacity is limited, valuation studies can help support the development of a legal framework or encourage natural resources protection enforcement capacity (see UNDP Equator Initiative Case Study Series - Community Mangrove Forest Conservation of Baan Bang La, Thailand, and Mikoko Pamoja, Kenya). By engaging closely with policy makers, valuation experts can ensure that their studies are tailored to decision-making needs with applied uses. Valuation studies are also more likely to be in demand and inform decision-making when there is high dependency or threat-driven urgency on the natural resources of concern. Therefore, capitalization of the opportunities from these external enabling conditions is critical to maximize the impact of valuation studies in informing decision makers (Waite et al., 2015), especially for countries like Malaysia with has traditionally prioritised economic development over conserving natural resources (Mokthsim and Osman Salleh, 2014).

Malaysia has the third highest mangrove extent globally (Hamilton and Casey, 2016) but experienced a mangrove area decline rate of approximately 793 ha per year (0.13%) between 1990 and 2017 (Omar et al., 2019). Much of the forest clearing was for urban development (e.g., infrastructure, housing) and economic development activities (e.g., commercial-scale agriculture and aquaculture). (Pourebrahim et al., 2011; Shahbudin et al., 2012). These destructive activities were also linked to exacerbation of coastal erosion and hardship faced by coastal poor (Hattam et al., 2020; Ruslan et al., 2022). While inevitable, the extent of mangrove destruction is arguably preventable to some extent. In a case study of the highly urbanized mangroves of Klang Islands, Peninsula Malaysia, Hattam et al. (2020) identified that private sector stakeholders have a low interest in, but high influence on local mangrove forests. Hattam et al. (2020) further noted that education and awareness raising of the importance of mangroves will be important for helping decision makers to reduce destructive activities. This suggests a role for valuation studies that can clearly articulate the importance of mangrove services and support cost benefit analyses. To date, there are considerable scientific studies examining the important services provided by Malaysian mangroves, such as their role in supporting complex food chains (Chong, 2005; Chew et al., 2012; Muhammad-Nor et al., 2019; Then and Chong, 2022), and the provision of nursery and habitat for fish, shrimps and birds species (Sasekumar and Chong, 1991; Norhayati et al., 2009; Chong et al., 2012). There is also a growing number of valuation studies that assess the ecosystem service values of Malaysian mangroves (e.g., Bann, 1999; Kaffashi et al., 2015; Hong et al., 2017; Hasan-Basri et al., 2020). However, based on available literature, there is no systematic compilation and assessment of these studies in terms of knowledge gaps and impact on decision making that would be important to help direct the future of ecosystem service valuation studies.

Therefore, this study aimed to (1) synthesize and compare valuation estimates of existing mangrove ecosystem services in Malaysia, (2) assess the effectiveness of mangrove ecosystem service valuation studies against a set of best practices, and (3) identify the gaps in developing functional and impactful valuation studies. To achieve these objectives, existing Malaysian mangrove ecosystem service valuation studies were collated and reviewed using a systematic literature review approach. Each study was then assessed against advocated criteria from global best practices and lessons learnt from other ecosystem service valuation studies for their effectiveness in informing decision making. This study does not critique the technical aspects of each method, but rather focuses on how they are applied, especially in relation to decision-making and stakeholder engagement. The challenges and opportunities of applying these best practices in Malaysia were discussed, with the overarching goal to advance and integrate ecosystem service valuation studies for improved mangrove decision making.

2 Methodology

2.1 Criteria of best practices of ecosystem service valuation and conditions to support its use in decision making

The first step was to identify and collate the criteria for best practices in conducting an effective ecosystem service valuation to inform policy and decision makers. We reviewed the following documents: de Groot et al. (2006); DEFRA (2007); Stelk and Christie (2014); Olander et al. (2015) and Schuster and Doerr (2015), which were selected for their applied nature, coverage of a range of valuation methods and specific detail relevant to wetland and coastal environments. Recommended best practices were collated to create a summary of best practices in valuing ecosystem services. Based on this review, five best practice criteria for implementing ecosystem service valuation studies were identified, which would serve as benchmarks to assess the effectiveness of ecosystem service valuation studies in Malaysia:

a. **Clear project goal(s) and policy question(s).** Identifying clear policy questions from the beginning will allow the researchers to determine the appropriate level of stakeholder engagement, appropriate valuation method and data needed (de Groot et al., 2006; Stelk and Christie, 2014; Schuster and Doerr, 2015). The policy question may be linked to the impacts of particular activities, the claims of specific stakeholders or a possible change in collective rules. For example, an ecosystem service valuation study by Cooper et al. (2009) raised awareness of the contribution of coral reefs and mangroves to the GDP of Belize, which then led the local government to enact new policies on fisheries, shipping and offshore oil drilling regulations. The use of ecosystem service valuation can be broadly categorised into three types: informative, decisive and technical (Laurans et al., 2013):

(1) Informative use: studies provide broad-based information that may indirectly influence decision making, for example *via* knowledge improvement and awareness-raising on importance of accounting for ecosystem services, providing justification and support; or merely introducing 'accounting indicators' for stakeholders or decision makers with which they may not be familiar. Green accounting indicators in the form of natural capital and environmental cost are vital information to assist in the management of environmental and operational costs of natural resources (Muralikrishna and Manickam, 2017);

(2) Decisive use: studies are designed to inform a specific decision, identifying impacts of specific scenarios that are economically relevant, physically quantifying impacts as benefits or costs, and then calculating a summary monetary valuation. A study of this type may project future effects of management interventions, comparison of management options, and facilitate trade-offs. In particular, environmental impact assessment value the likely ecological cost of a proposed project or development (MacKinnon et al., 2018);

(3) Technical use: this involves cases where ecosystem service valuation is applied after choosing a policy or project to adjust the economic instrument that will implement the decision. For instance, a study was established to calculate damage compensation after environmental degradation or price setting on certain ecosystem services.

b. **Strong engagement with all relevant stakeholders/ decision makers.** Identification of important stakeholders groups that will be affected by any changes in management as a result of the ecosystem service valuation study is critical (DEFRA, 2007; Olander et al., 2015). These include decision makers (e.g., landowners, local government, and policymakers) and beneficiaries and detrimentally-impacted end users (e.g., local and adjacent residents, business owners using the lands, visitors) and ecosystem advocates (e.g. environmental NGOs and other civil society groups). Following identification, strong stakeholder engagement throughout the valuation process is required to produce an appropriate study design, enable effective data collection, determine legitimacy and credibility of results, and to support capacity building (Brown et al., 2001; de Groot et al., 2006). A strong stakeholder engagement is typically indicated by extensive use of stakeholder analysis tools, involving wider group of stakeholders or by subjecting the process of public reviews (Waite et al., 2015; Raum, 2018; Hattam et al., 2020).

c. **Clear causal link(s) between ecosystem services and socio-economic variables.** Identifying and connecting the causal links between drivers of ecosystem change, ecosystem health, ecosystem services, and resource users is essential for stakeholders and decision makers (Olander et al., 2015). A detailed description or illustration of a causal chain and relationships can help garner support of stakeholder groups towards suggestions made by valuation practitioners. Demonstrating a causal link in ecosystem service valuation can sometimes help identify potential equity issues and other often overlooked factors. For example, several connections between cultural ecosystem services (such as urban green spaces) and social determinants of health (such as economic stability and social capital) were demonstrated by Zelenski et al. (2015) and Jennings et al. (2016).

d. **Relevant choice of valuation methodology, indicators, metrics and measurements.** Various valuation methodologies and measurements can be used to value ecosystem services, such as revealed preference (market price, travel cost), stated preference (contingent valuation, choice experiments) and benefit transfer. Each method is appropriate to specific types of ecosystem services and policy questions. For example, market prices can be used for ecosystem services that are traded through markets (e.g. for provisioning services), stated preference methods are particularly useful for capturing non-market values (e.g. for regulating and cultural services), while benefit transfer is useful in data poor situations and can draw on studies from other locations relevant to all ecosystem services (see National Research Council (2005) and Barbier (2007) for details of these methods). It is important to note that valuation methodologies are not necessarily mutually exclusive and more than one method can be applicable for a given policy question. For example, a combination of survey data from actual recreational usage patterns of a site (i.e., through revealed preferences) and from anticipated changes to those patterns under hypothetical increases in trip costs (i.e., through stated preferences) could reduce hypothetical bias, and provide more accurate valuation estimates (Haipeng and Xuxuan, 2012; de Corte et al., 2021). Valuation practitioners should choose appropriate strategies that best answer the policy question and provide tailored results that are appropriate to relevant stakeholders.

e. **Effective dissemination and communication of results with stakeholders/decision makers.** Following valuation studies, strategic dissemination of results and policy recommendations are crucial to ensure that the decision makers and stakeholders are well informed for decision making (de Groot et al., 2006; Olander et al., 2015; Waite et al., 2015). A well-developed communication and outreach strategy, drawing on diverse media platforms such as traditional and social media, can help with both widespread and targeted communication of results. Bundling the valuation results according to the interests of target stakeholders can increase the likelihood that the valuation results being used and relevant locally. In addition, standardisation in reporting valuation outcomes can increase the credibility and comparability of studies (Boyd and Banzhaf, 2007; de Groot et al., 2012; Seppelt et al., 2012).

2.2 Literature review and assessment of Malaysia ecosystem service valuation studies

Existing Malaysian mangrove ecosystem service valuation studies were identified and collated for systematic review following the Preferred Reporting Items for Systematic Reviews and Meta-Analysed (PRISMA) method. Relevant articles were identified from the Web of Science (WOS) and SCOPUS databases using the following search criteria: (1) (mangrove*) AND ("ecosystem* servic*") AND (valu*) AND (Malaysia); (2) (mangrove*) AND (economic) AND (valu*) AND (Malaysia); and (3) (mangrove*) AND ("benefit transfer" OR "avoided cost" OR "conversion cost" OR "damage cost" OR "mitigation cost" OR "opportunity cost" OR "replacement cost" OR "restoration cost" OR "bio-economic modelling" OR "factor income" OR "production function" OR "consumer surplus" OR "hedonic pricing" OR "market price" OR "net price method" OR "public investments" OR "substitute goods" OR "travel cost method" OR "choice modelling" OR "contingent ranking" OR "contingent valuation" OR "participatory valuation") AND (Malaysia). After removing duplicates, a total of 184 articles were identified. Reports were then screened and filtered, retaining only articles documenting a monetary value for mangrove ecosystem services. Publications that value mangrove ecosystem services for an undefined geographical location or did not clearly document the valuation methodology were excluded. In addition, grey literature documenting mangrove ecosystem service valuation studies in the form of reports, articles and dissertations were obtained from local libraries and relevant government ministries' archives. This whole exercise resulted in a total of 17 publications for the following data extraction and assessment.

All 17 publications were reviewed and qualitative and quantitative data relevant for comparison across studies were extracted. For each publication the publication year, type of publication, geographic location of the study, valuation methodology, estimated mangrove ecosystem service values and units were extracted and tabulated. Ecosystem service values were organised into categories based on the classification scheme by TEEB (2010). Based on the procedure described in the TEEB database (TEEB, 2010), all values were standardised into USD value on the nasis of Purchasing Power Parity in year 2007 that allowed for direct comparisons between collated studies and global estimates from de Groot et al. (2012); Costanza et al. (2014) and Himes-Cornell et al. (2018b). Where similar units were used, values were pooled to obtain an average. Subsequently, all studies were further assessed against the best practice criteria for ecosystem service valuation.

The studies were scrutinised for basic information including the role of valuation practitioners, the primary use of ecosystem service valuation, main policy question, type of stakeholders engaged and stakeholder engagement, ecosystem service valuation methodology and result dissemination strategy employed. We adopted the typology of stakeholders according to Raum (2018), where (i) producers were defined as those stakeholders who produce goods or services through particular ecosystem services; (ii) users are the stakeholders who passively use or benefit from the use of particular ecosystem services; (iii) regulators are those stakeholders with the ability to set either formal or informal rules to govern the actions of other stakeholders about ecosystem services; (iv) researchers were defined as any stakeholder which engages in scientific research and understanding, including modelling, but excluding monitoring and observing; and (v) monitors are the stakeholders who engage in scientific monitoring and observing of particular ecosystem services, and inform other stakeholders. The whole text was then examined for explicit mentions of links between stakeholders and ecosystem services. For research outcomes and dissemination, texts were examined for description of dissemination, communication or outreach activities. Peer-reviewed studies were also scrutinised in terms of the journal impact factors and the number of citations. For valuation studies that aimed to 'determine appropriate charging rates for environmental use' in conservation areas (e.g., visitor entry fees), changes in visitor entry fees post valuation studies were examined *via* internet search and personal communications from residents.

3 Results

3.1 Overview of ecosystem service valuation studies

A total of 184 publications were screened and 17 publications and reports were identified for inclusion in this study. These studies reported values of ecosystem services produced by mangrove forests (see Supp. Appendix 1 for complete list of publications). Tables 1, 2 summarise these 17 publications which covered 10 of the 13 states of Malaysia (Figure 1). Perak state has the greatest number of mangrove ecosystem service valuation studies (4), with all studies focused on the Matang Mangrove Forest Reserve. No documented mangrove ecosystem service valuations were found for the states of Pahang, Negeri Sembilan and Malacca despite the known presence of mangroves in these states. Among the analysed valuation studies, five estimated the value of mangroves as a whole without indicating specific types of ecosystem services. The remainder valued nine specified types of ecosystem services, with tourism (including recreational) services being captured most frequently (10 studies), followed by carbon sequestration services (7), fisheries production services (4), coastal protection services, including storm surge protection (3), and other services including timber production, non-timber forest production, aquaculture production, riverine production and water quality improvements services. Only three studies were conducted before the year 2000, i.e., in years 1992, 1994 and 1999, two studies in the subsequent decade (i.e., in 2009), and the remaining 12 studies conducted between 2011 to 2020.

Market price was the predominant valuation methodology (16 estimates) used for direct use services such as fisheries, non-timber forest production and carbon sequestration (Figure 2). The second most dominant valuation methodology was contingent valuation (12 estimates), mainly to estimate tourism and recreational cultural services (6 estimates), and one each for fisheries productions and coastal protection service. Four studies estimated the total value of mangroves using contingent valuation without specifying the types of ecosystem services (willingness to pay for mangrove preservation). The travel cost and replacement cost methods were less commonly used. The former was used for three tourism cultural services and the latter for two coastal protection regulating services. Only one study used the benefit transfer to estimate the value of water quality improvement. Value for water quality treatment in this study was estimated based on a meta-analysis of global mangrove ecosystem services' economic value by Salem and Mercer (2012).

Due to the high variety of measurement units and valuation methodologies, the value estimates for each state and type of service are not directly comparable. Following standardization of estimated valuation to 2007 USD rates to maintain parity, the mean value of local mangrove services valuation was compared against collated global estimates of mangrove ecosystem services value (Table 3). Estimates of food (fisheries production and aquaculture), water and erosion prevention services in Malaysian mangroves appeared to be higher than synthesis from global

TABLE 1 Summary of mangrove ecosystem service valuation in Malaysia.

Components	Valuation methods	Estimated value (USD)	Units	Location	State	Study period	References
TEV Preservation value	CV	44,408	$Hectare^{-1}$ $year^{-1}$	Kuala Selangor Mangrove Forest	Selangor	1994	Leong et al., 2005
Mangrove value	CV	3	per year	Kuching Delta Mangrove Forest	Sarawak	2012	Shuib et al., 2012
Mangrove value	ES	307	$Hectare^{-1}$	Sungai Merbok Forest Reserve	Kedah	2013	Khuzaimah et al., 2013
Mangrove value	CV	3	per year	Matang Mangrove Forest Reserve	Perak	2017	Ramli et al., 2017
Mangrove value	CV	3,252	$Hectare^{-1}$	Kampung Sungai Melayu	Johor	2018	Sunoto et al., 2020
Mangrove value	CV	18,587	$Hectare^{-1}$ $year^{-1}$	Kuala Perlis Mangrove	Perlis	2020	Hasan-Basri et al., 2020
Aquaculture production	MP	10,479	$Hectare^{-1}$ $year^{-1}$	Kuala Selangor Mangrove Forest	Selangor	1994	Leong et al., 2005
Fisheries production	MP	72,396,170	$Hectare^{-1}$	Sarawak Mangrove Forest Reserve	Sarawak	1992	Bennet and Reynolds, 1993
Fisheries production	MP	6,605	$Hectare^{-1}$ $year^{-1}$	Kuala Selangor Mangrove Forest	Selangor	1994	Leong et al., 2005
Fisheries production	CV	835	$Hectare^{-1}$ $year^{-1}$	Benut	Johor	1999	Bann, 1999
Fisheries production	MP	18,292	$Hectare^{-1}$ $year^{-1}$	Teluk Air Tawar-Kuala Muda coast	Penang	2016	Foong et al., 2016
Fisheries production	MP	413	$year^{-1}$	Kudat	Sabah	2016	Mojiol et al., 2016
Non-timber forest product	MP	135	$Hectare^{-1}$ $year^{-1}$	Kuala Selangor Mangrove Forest	Selangor	1994	Leong et al., 2005
Timber production	MP	422,770	$Hectare^{-1}$	Sarawak Mangrove Forest Reserve	Sarawak	1992	Bennet and Reynolds, 1993
Timber production	MP	98	$Hectare^{-1}$	Matang Mangrove Forest Reserve	Perak	2015	Aziz et al., 2015
Carbon sequestration	MP	197	$Hectare^{-1}$	Matang Mangrove Forest Reserve	Perak	2015	Aziz et al., 2015
Carbon sequestration	MP	105,525	$Hectare^{-1}$	Teluk Air Tawar-Kuala Muda coast	Penang	2016	Foong et al., 2016
Carbon sequestration	MP	5,191[a]	$Hectare^{-1}$	Kuala Selangor Nature Park	Selangor	2017	Hong et al., 2017
Carbon sequestration	MP	3,211[a]	$Hectare^{-1}$	Sungai Haji Dorani	Selangor	2017	Hong et al., 2017
Carbon sequestration	MP	16,593[b]	$Hectare^{-1}$	Kuala Selangor Nature Park	Selangor	2017	Hong et al., 2017
Carbon sequestration	MP	10,263[b]	$Hectare^{-1}$	Sungai Haji Dorani	Selangor	2017	Hong et al., 2017
Coastal protection	RC	16,630	$Hectare^{-1}$ $year^{-1}$	Kuala Selangor Mangrove Forest	Selangor	1994	Leong et al., 2005
Coastal protection	CV	1,342	$Hectare^{-1}$ $year^{-1}$	Benut	Johor	1999	Bann, 1999
Coastal protection	RC	3,004	$Hectare^{-1}$ $year^{-1}$	Teluk Air Tawar-Kuala Muda coast	Penang	2016	Foong et al., 2016
Riverine production	MP	46	$Hectare^{-1}$ $year^{-1}$	Kuala Selangor Mangrove Forest	Selangor	1994	Leong et al., 2005
Water quality improvement	BT	4,577	$Hectare^{-1}$ $year^{-1}$	Teluk Air Tawar-Kuala Muda coast	Penang	2016	Foong et al., 2016
Tourism	MP	12,935,237	$Hectare^{-1}$	Sarawak Mangrove Forest Reserve	Sarawak	1992	Bennet and Reynolds, 1993
Tourism	TC	1,211	$Hectare^{-1}$ $year^{-1}$	Kuala Selangor Mangrove Forest	Selangor	1994	Leong et al., 2005
Tourism	CV	5	$Hectare^{-1}$ $year^{-1}$	Benut	Johor	1999	Bann, 1999

(Continued)

TABLE 1 Continued

Components	Valuation methods	Estimated value (USD)	Units	Location	State	Study period	References
Tourism	CV	10	Visitor^{-1} year^{-1}	Matang Mangrove Forest Reserve	Perak	2009	Ahmad, 2009
Tourism	CV	2	Visitor^{-1} year^{-1}	Pulau Redang Marine Park (PRMP)	Terengganu	2009	Yakob et al., 2009
Tourism	CV	2	Visitor^{-1} year^{-1}	Pulau Payar Marine Park (PPMP)	Kedah	2009	Yakob et al., 2009
Tourism	CV	419	Hectare^{-1} year^{-1}	Penang National Park	Penang	2012	Kaffashi et al., 2015
Tourism	TC	34	Hectare^{-1}	Matang Mangrove Forest Reserve	Perak	2015	Razak et al., 2018
Tourism	CV	3,706	Hectare^{-1} year^{-1}	Teluk Air Tawar-Kuala Muda coast	Penang	2016	Foong et al., 2016
Tourism	TC	6,543	Hectare^{-1}	Kilim Karst Geoforest Park	Kedah	2019	Matthew et al., 2019

All values were standardized to year 2007 estimates. ES, Ecosystem service valuation method (remote sensing); CV, Contingent valuation; MP, Market price; BFT, Benefit function transfer; TC, Travel cost; TEV, Total Economic Value.
Estimated value of carbon reported by Hong et al., 2017 are in terms of voluntary market price (a) and from regulated market European Union Emissions Trading System (b).

TABLE 2 Mangrove extent (ha) for each state in Malaysia (2017), and their corresponding number of ecosystem service valuation (ESV) studies up to 2020.

State	Total mangrove area 2017 (ha)	Total number of ESV studies
Perlis	49	1
Kedah	7,725	3*
Penang	1,967	2
Perak	44,990	4
Selangor	20,853	2
Negeri Sembilan	1,557	0
Melaka	1,241	0
Johor	26,818	2
Pahang	3,759	0
Terengganu	1,571	1*
Kelantan	422	0
Peninsular Malaysia	110,952	14*
Sabah	378,195	1
Sarawak	139,890	2
Grand total	629,037	17*

Mangrove area data were collated from Omar et al. (2019). *Valuation study by Yakob et al., (2009) covered two Malaysian states (Kedah and Terengganu).

estimates. Meanwhile, the recreation and tourism ecosystem services in Malaysian mangroves were valued lower compared to global estimates.

3.2 Assessment of studies against criteria of best practices

3.2.1 Ecosystem service valuation study background

Out of the 17 ecosystem service valuation studies, the majority of the identified valuation practitioners were from academic and scientific institutions (82.4%). Only two studies were undertaken by government agencies (state forestry departments) and one by a non-government organization (Table 4). Although there were apparent collaborations between local universities (with inclusion of foreign universities in a few studies) in conducting the valuation, there were no apparent or strong collaborations between the universities and government agencies, who are often the main decision makers in Malaysia in terms of mangrove management. The lack of cross-agency collaborations was also seen for the three studies conducted by government agencies and non-government agencies, who appeared to carry out the valuation independently.

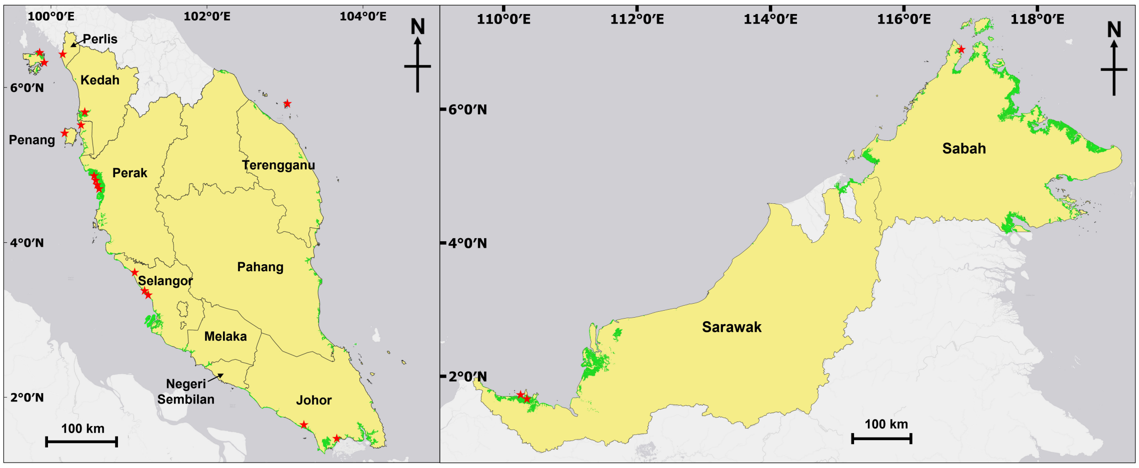


FIGURE 1
Map of Malaysia. Red stars indicate valuation study sites of collated mangrove ecosystem service valuation studies in Malaysia. Green patches overlaying the map indicate mangrove forest coverage (dataset from Bunting et al., 2018).

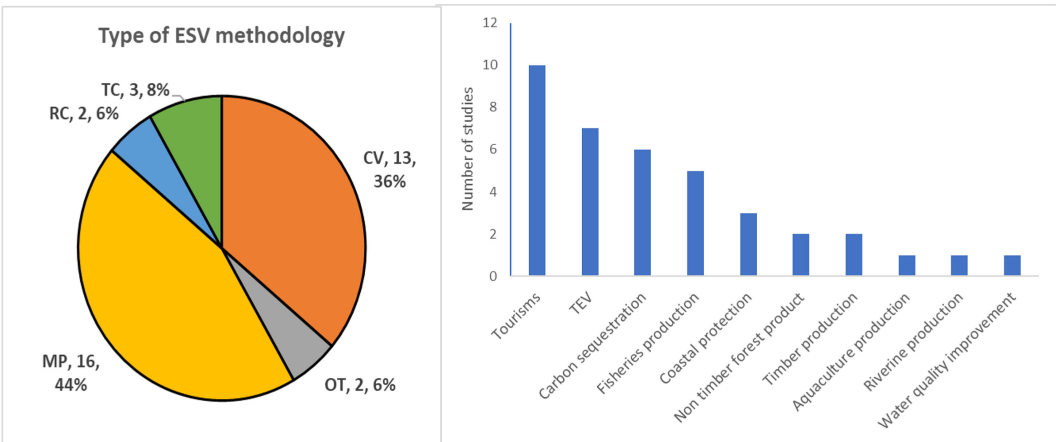


FIGURE 2
Summary of ecosystem service valuation (ESV) methodologies and ecosystem services that had been valued in Malaysia. CV, contingent valuation; MP, market price; BT, benefit transfer; RC, replacement cost; TC, travel cost; OT, others including the benefit transfer and ecosystem service valuation method (remote sensing).

3.2.2 Identification of a clear project goal, policy question, boundaries and scope

Of the 17 ecosystem service valuation studies assessed, only two types of valuation uses were identified (Table 4). Twelve of these studies were classified as ecosystem service valuation for informative use, while the remainder were for decisive uses. All but one of the informative use valuation studies were conducted with the main purpose to raise awareness of the value of mangroves, the exception was a study that aimed to provide justification for and support to certain decision making. For the decisive use valuation studies, two were conducted to help determine charging rates for environmental use, while the other three aimed to inform decision making by comparing costs and benefits of different uses of the environment and assessing trade-offs.

Among the 11 valuation studies designed for informative use, i.e., to raise awareness of the value of mangrove ecosystem services, eight of them addressed specific types of mangrove ecosystem services to their respective stakeholders. Specifically, these studies related the ecosystem services to the end-users and decision makers surrounding the mangrove forest. For instance, Bennet and Reynolds (1993) noted that local residents depended

TABLE 3 Comparisons of the mean estimated value of ecosystem services in Malaysia with global data.

Reference study(s)	de Groot et al. (2012); Costanza et al. (2014)	Himes-Cowell et al. (2018b)	Synthesis from this study
Reference ecosystem(s)	Coastal wetlands (tidal marsh, mangroves, and saltwater wetlands)	Mangroves	Mangroves
Ecosystem services category	Mean value across studies	Mean value across studies	Mean value across studies (number of studies; min-max value)
Food	1,111	8,319	9,053 (3; 835 – 18,292)
Water	1,217	799	2,312 (2; 46 – 4,577)
Climate regulation	65	34,756	23,497 (6; 197 – 105,525)
Erosion prevention	3,929	930	6,992 (3; 1,342 – 16,630)
Recreation and tourism	2,193	3,526	1,335 (4; 5 – 3,706)

To maintain parity with other studies, all values were standardized to 2007 and USD per hectare per year.

heavily on the mangrove resources (fisheries and timber production) and the gain and loss of tourism services from losing the mangrove forest to oil palm plantations and aquaculture practitioners instead of conserving the forest. Hong et al. (2017) noted that the amount of carbon stocks able to be sequestered by the mangrove and the potential revenue in carbon stock trading using the market price for the mangrove manager. On the other hand, three studies valued the total economic value of mangroves conservation, but without clearly specifying the types of ecosystem services (Shuib et al., 2012; Hasan-Basri et al., 2020; Sunoto et al., 2020); these studies did however given information on the role and usage of mangrove users by percentage. For example, in the Kuala Perlis mangrove forest (Hasan-Basri et al., 2020), a majority of the users of mangroves were fishermen (82%) and fish-cage workers (13%). Meanwhile, Sunoto et al. (2020) noted that 67.6% of the villagers (from Kampung Sungai Melayu) were dependent on local mangrove resources (fisheries activities) for their livelihood.

Bann (1999) is the only study designed to provide justification and support to specific decision making. It aimed to inform the Johor State Forestry Department on whether to change the status of Benut mangrove forest from state land into a permanent reserve forest. This study employed contingent valuation to estimate the demand for public services, and the economic value of environmental change.

Three decisive use valuation studies were conducted to determine appropriate charging rates for access to local mangrove forests (Yakob et al., 2009; Kaffashi et al., 2015; Ramli et al., 2017). These studies focused on the cultural ecosystem services: tourism and recreational services using the contingent valuation approach, and the end-user willingness to pay (local residents and tourists) for mangrove preservation as ecosystem service value indicators. For all studies, the visitors to the mangrove forest or park were notably able to appreciate the existence of the mangrove. Meanwhile, the local mangrove managers indicated they were able to accrue funds from increased entrance fees that could be used to support better management.

Two valuation studies performed cost-benefit analyses of different uses of mangrove forest (Aziz et al., 2015; Foong et al., 2016). These studies covered at least two types of mangrove ecosystem services for valuation. Foong et al. (2016) estimated the value of multiple mangrove ecosystem services as beneficial to both the end-users (local residents and fishermen) and decision makers (mangrove managers) under different mangrove management regimes (intact mangrove forest vs. extensive aquaculture farm) *via* benefit transfer (Table 1). Aziz et al. (2015) examined the cost-benefit of different management scenarios of mangrove timber to the mangrove manager. The economic value of timber production and carbon stocks became the indicators and valuation units for the mangrove managers.

Most of the studies (64.7%) were able to illustrate a direct causal link between mangrove ecosystem services, stakeholders and valuation outcomes. For example, links were demonstrated between local mangrove forests and economic importance from tourism (Yakob et al., 2009; Kaffashi et al., 2015; Ramli et al., 2017; Matthew et al., 2019), and between various mangrove resources with the livelihoods of local residents (Ahmad, 2009; Shuib et al., 2012; Mojiol et al., 2016; Sunoto et al., 2020). Foong et al. (2016) provided detailed causal linkages between mangroves and adjacent mudflat ecosystem services to the residents and fishers living close to the mangrove and made a connection to the aquaculture project as well as a cost-benefit comparison between different management scenarios for all involved stakeholders. Aziz et al. (2015) created a link between mangrove conservation with timber extraction and the carbon market. The multiple levels of jurisdiction, stakeholders, opportunity cost and assumption were clearly defined in this study.

3.2.3 Identification and strong engagement with stakeholders/decision makers

In terms of identification of and engagement with relevant stakeholders over the course of the ecosystem service valuation process, the 17 studies collectively identified seven groups of stakeholders (Table 4). These include the residents adjacent to

TABLE 4 Summary of Malaysia ecosystem service valuation (ESV) assessment based on best practices criteria.

Criteria	Number of studies	Percentage
ESV study background		
ESV study practitioner role:		
Universities	14	82
Government agencies	2	12
NGO	1	6
Identification of a clear project goal, policy question, boundaries and scope		
Type of ESV uses		
Informatic use	12	71
Decisive use	5	29
Technical use	0	0
Objective and policy question of ESV		
Raise awareness of the value of ecosystems	11	65
Provide justification and support to certain decision making	1	6
Determine appropriate charging rates for environmental use	3	18
Compare costs and benefits of different uses of the environment and assess trade-offs	2	12
Identification and strong engagement with stakeholders/decision makers		
Identified major stakeholder groups (with types in bracket)		
Residents adjacent to mangrove (users)	12	71
Fishermen (producers, users)	9	53
Tourists (users)	9	53
Aquaculturists (producers, users)	3	18
Plantation manager (producers, regulators)	1	6
Local forestry department (regulators)	8	47
Mangrove manager (regulators)	9	53
Stakeholders' engagement		
Yes	12	71
No	5	12
Type of direct stakeholders' engagement		
Engagement during design stage	8	47
Engagement during implementation and analysis stage	13	76
Engagement after valuation study	2	12
Effective results dissemination and communication with stakeholders/decision makers		
Publication in the scientific journal	13	77
Malaysian journal	4	24
International journal	9	53
Publication in the grey literature (book/technical report/case studies)	4	24

the mangrove forest (70.6%), fishermen (52.9%) and tourists (52.9%) as 'users' stakeholders, and the aquaculturists (17.6%), plantation managers (5.9%), local forestry department (47.1%) and mangrove managers (52.9%) as stakeholders having stronger control over the governance of mangrove forest (producers and regulators). The majority of the studies indicated engagement with stakeholders (n = 15; 88.2%) while the rest did not. Among the 15 studies that included stakeholder engagement, 47.1% of the studies had engaged stakeholders during the design stage of valuation, 76.5% had direct stakeholder engagement in implementation and analysis stages, and only two studies (11.8%) indicated stakeholder engagement beyond the completion of the valuation studies. Engagement with stakeholders during the design stage of valuation studies was mainly with the local forestry department in the form of acknowledgement and endorsement of the projects, while only one valuation study engaged with local residents' representatives. Meanwhile, 12 valuation studies only engaged with stakeholders as the target audience for their contingent valuation and travel cost valuation studies (i.e., through questionnaire completion and interview as part of data analysis). The remaining two studies showed some degree of wider stakeholder engagement: Bann (1999) is a study conducted by the state forestry department suggesting communication between the study team and the wider organisation; whereas Foong et al. (2016) indicated exchange with the Forestry Research Institute of Malaysia, the Forestry Department and several local Non-Government Organizations.

3.2.4 Effective results dissemination and communication with stakeholders/ decision makers

For dissemination of valuation results to relevant stakeholders, the majority of the valuation studies were published in scientific journals (76.5%). Eight of the studies were published in international journals such as Forests (Impact factor, IF = 2.634), Ecological Economics (IF = 5.389), and Journal of Tropical Forest Science (IF = 0.770). The other five studies were published in Malaysian-based peer-reviewed journals, namely Journal of Tropical Resources and Sustainable Sciences, the Malaysian Journal of Economics, Planning Malaysia: Journal of the Malaysian Institute of Planners and The Malaysian Forester. In terms of citations recorded by ResearchGate, the number of citations for each study at the time of writing ranged from 2 – 144. Only one valuation study did not have information in numbers of citations. For the grey literature, two studies were published as technical reports for the purpose of informing specific stakeholders on mangrove management, and two studies in the form of book chapter and conference publication. Meanwhile, no studies have indicated or described valuation output dissemination in their texts.

3.3 Synthesis of Malaysian mangrove ecosystem service valuation studies

Drawing on the studies reviewed, key shortcomings in mangrove ecosystem service valuation in Malaysia are identified as follows:

a. The valuation units are not standardised across the whole country, even for the same services. There are variations among the valuation units used by different Malaysian valuation practitioners to value mangrove ecosystem services. This was exhibited particularly in the valuation of tourism cultural ecosystem services by several ecosystem service valuation studies (Table 1). Valuation units include value per hectare per year, value per visitor per year and value per hectare, all derived through the contingent valuation method. These values are not interchangeable, rendering comparative assessment impossible.

b. The majority (64.7%) of the studies reviewed make only a cursory reference to the potential use of ecosystem service valuation: Specifically, most of the authors merely indicated how the economic valuations of respective services could be used. They fail to describe how they could contribute to policy decisions or practical management. Furthermore, most were piecemeal studies which only evaluated one or a few ecosystem services, and with relatively simple causal links between ecosystem services and the stakeholders. Consequently, valuation in most cases is incomplete and not sufficiently relevant to inform socially optimal decisions. The take home message from most of the valuation studies to their intended stakeholders was a generic suggestion to value more highly the studied ecosystem services.

c. Most of the studies document limited or no clear collaboration between the valuation practitioners and relevant stakeholder groups, including decision makers. Eight out of 17 studies have identified specific and relevant stakeholder groups for their studies, such as the forestry department of the respective state and the local mangrove forest managers. However, engagements with these regulator stakeholders were limited to acknowledgement of permits approved by the local forestry department or mangrove manager to conduct research in mangrove forest (Hong et al., 2017; Hasan-Basri et al., 2020), or to providing valuation information to the regulator stakeholders (Yakob et al., 2009; Shuib et al., 2012; Aziz et al., 2015; Kaffashi et al., 2015). These academic studies reported limited involvement from other stakeholder institutions or with regulators.

d. There was limited documentation concerning valuation outcomes. Most of the valuation studies were published in a scientific journal, some with a high number of citations. However, there is no clear indication that decision makers use the said publications to support the drafting of new mangrove management policies, or revision of existing ones. For example, the valuation studies of Matang Mangrove Forest Reserve, Perak by Ramli et al. (2017), Penang National Park by Kaffashi et al. (2015), and Pulau Payar and Pulau Redang by Yakob et al. (2009) were designed to determine appropriate charging rates for the local mangrove forests. However, they appear to have had no impact on the mangrove managers and the rates charges, specifically no evident changes in the park entrance fees to date (personal comm. with park managers).

Only two valuation studies appeared to have successfully informed the valuation outcome (i.e., been used or acknowledged by decision-makers in some way). The valuation study by Bann (1999) was used to inform the decision to change Benut mangrove forest from a state land forest to permanent forest reserve. As seen from the Summary of the State of Johor Forest Management Plan for the Period Between 2006-2015, the forest was subsequently gazetted as a permanent forest since 2005. The study by Foong et al. (2016) appeared to garner attention in later years, with an open public talk that was held in 2019 in conjunction with World Wetland Day, re-emphasizing the status of Teluk Air Tawar-Kuala Muda coast as an Important Bird and Biodiversity Area.

4 Discussion

From the five best practice criteria identified for ecosystem service valuation, we found that the limited studies in Malaysia were generally sound in terms of methodology and scope but lacking in terms of key stakeholders' connections and output dissemination. These limitations appeared to have reduced the effectiveness of the studies in terms of uptake of results by decision makers and buy-in from other stakeholder groups. Here, we summarize the identified gaps and discuss the key opportunities and practical way forward for future ecosystem service valuation and broader ecosystem assessment efforts by linking to relevant national policies and international commitments. These recommendations draw from lessons learnt from other case studies outside of Malaysia and are broadly applicable in the Southeast Asian region and for valuation of mangroves and other similar coastal habitats.

4.1 Increased connections/engagements between valuation practitioners and key stakeholder groups

One key best practice of ecosystem service valuation is the importance of stakeholder identification and engagement (Barbier, 2007; Waite et al., 2015; Raum, 2018). Many previous efforts to manage the environment and natural resources in Malaysia and elsewhere were not highly successful due to inadequate consideration given to various stakeholders involved (including their potentially conflicting interests and perspectives) by policymakers or local planners (Grimble et al., 1994; Waite et al., 2015; Marre et al., 2016). Given that values are context and time-specific, the value for different stakeholder groups or communities placed on ecosystem services can vary considerably. Stakeholder analysis is therefore a key practical step to help identify and understand stakeholders: how they are affected by ecosystem services, how they influence them, and their role in (public) decision making (Renard, 2004). Insights into the range of values associated with specific ecosystem benefits held by different stakeholders can in turn be used to support more effective and equitable engagement, and to inform valuation design and delivery, thus enabling informed decision making (Marre et al., 2016).

In the context of forest ecosystem services, crucial stakeholders often include government organizations as regulators; producers who extract forest goods and services; and users who use or benefit from mangrove ecosystem services. In Malaysia, mangrove forest regulators are typically top-down, centralised, and compartmentalised (Hattam et al., 2020). Communication and coordination between different departments and tiers of government are complicated, thereby rendering the mangrove management fragmented and poorly integrated with land-use policy directions (Friess et al., 2016; Amir, 2018). On the other hand, other important stakeholders such as local communities have strong interest in mangroves but are often powerless to affect change (Hattam et al., 2020). In this context, local valuation practitioners should execute stakeholder mapping early in the study design process and include highly influential local state agencies in engagement activities to ensure just, equitable decision making. Incorporating stakeholder-driven scenarios in ecosystem service valuation design can help ensure that the valuations are aligned with the problem statements by decision-makers (Henrichs et al., 2010; McKenzie et al., 2011) and allow for contrasts in gains and losses to ecosystem services for determining win–win solutions (Barnett et al., 2016; Rau et al., 2020).

4.2 Broadened portfolio of result dissemination platforms for ecosystem service valuation studies

Understanding the influence of the studies assessed in this review has been challenging due to limited evidence. The dissemination of valuation findings is essential for ensuring they are accessible for use in decision making (de Groot et al., 2006), but academic publishing of findings alone is no longer sufficient to ensure research use (Ament, 1994). Publication citation rate indicates some level of study uptake, but is somewhat controversial (Seglen, 1989; Cagan, 2013) and does not necessarily verify the solidity and societal value (Aksnes et al., 2019). On the other hand, ecosystem service valuation studies from grey literature, such as technical reports and case studies, may potentially have wider reach, especially to the decision makers. They are context specific and may contain relevant information for decision makers that are not usually captured by peer-reviewed literature (Rothstein and Hopewell, 2009). Valuation studies by Bann (1999) and Foong et al. (2016) were grey literature article and not published in peer-reviewed journal articles but appeared to have at least successfully informed specific groups of stakeholders.

The lack of uptake of ecosystem service valuation studies may be attributed to at least two barriers, i.e., the research evidence is not available in an accessible format for the policymaker and the evidence is disregarded due to clashes in political or ideological reasons (Hawkins and Parkhurst, 2016; Uzochukwu et al., 2016). To address the first barrier, diversification of strategies using suitable platforms to disseminate valuation outcome is needed (Avishek et al., 2012). A policy brief, i.e., a short document synthesizing the results of one or multiple studies, is one strategy to promote the use of research (Arnautu and Dagenais, 2021) as well as having more effective science-to-policy dialogues that is free from structural or political barriers (Jones et al., 2008; Young et al., 2014). Significance of policy briefs and science-to-policy dialogues was often recognized in the public health sector (Suter and Armitage, 2011; Kilpatrick et al., 2015; Damani et al., 2016; Nabyonga-Orem et al., 2016). Through these mediums of exchange, valuation practitioners can bring ecosystem service valuation results to policymakers and may gain feedback on how to tailor valuation approaches to meet their needs. To address the second barriers of political or ideological differences between conservation and development (Wiesmann et al., 2005; Apostolopoulou and Pantis, 2010; Scoones, 2016), valuation teams should be transdisciplinary in composition to include economists, political, social, communications, and natural scientists (Costanza and Kubiszewski, 2012; Schneider et al., 2019). The transdisciplinary approach may promote understanding of the realities of evidence-based research and policy making within the team and improve communication outreach to different stakeholder groups. Additionally, the team should identify local champions that are well versed with ecosystem services that can help to communicate valuation outcomes (Cooper et al., 2009; Waite et al., 2015). These individuals or organisations often have established platforms or communication tools to garner support from influential groups to help sway the opposing political stance, and leverage needed change. Working with these local champions for broad result dissemination will likely increase buy-in from key strategic figures including local communities adjacent to affected mangroves, and influence decision making processes. This is evident from some case studies in Indonesia, where local champion successfully empowered local communities in implementing climate change adaptations (Septiarani and Handayani, 2016) and poverty alleviation efforts (Tranggono et al., 2021).

4.3 Congruence within local valuation studies, and with global valuation standards

Due to the complexity of ecosystem services assessment and the nature of policy questions, the metrics employed within each valuation study can be very different from others, thus rendering them incomparable. Lack of comparability translates into difficulties for decision makers or other valuation practitioners in facilitating direct comparison between sites or in transferring values from studied sites to new sites of interest. Aside from comparison within countries, standardization in the framework and reporting of ecosystem service valuation among countries is also crucial, especially the identification of beneficiaries of ecosystem services at different scales (Boyd and Banzhaf, 2007; de Groot et al., 2012) and in facilitating transboundary ecosystem services assessment (Dang et al., 2021). While the units reported may reflect the valuation question being asked, future valuation studies should report values in a range of units where possible to aid study comparability.

One significant global comparative effort is the development of the Ecosystem Services Valuation Database (ESVD) for the study of 'The Economics of Ecosystems and Biodiversity" (TEEB, 2010). The database hosts at least 6,700 value records from over 950 studies globally (Foundation for Sustainable Development 2021), thus supporting the ease of value transfer applications and meta-analysis across multiple studies. While the adopted ecosystem service classification systems in ESVD, i.e., the TEEB classification (TEEB, 2010) and the CICES V5.1 classification (Haines-Young and Potschin, 2018), may require review and adaptation to suit local contexts, there are clear grounds and overall benefits from employing such a global standard for standardization of spatial and temporal units of the ecosystem services. By adopting widely agreed-upon standards of best practices and reporting, the quality and comparability of valuation results can be improved.

4.4 Evolving national policy landscapes for ecosystem services assessment and opportunities

It is recognised that valuation studies are more likely to be accepted or able to inform the decision makers if the ecosystem services being valued are of high importance to the key stakeholders (Waite et al., 2015; Marre et al., 2016). In the context of Malaysia, the importance of ecosystem services and marine goods from intact mangroves are well recognized particularly after the 2004 Indian Ocean tsunami (Asma et al., 2012). This particular disaster had also been identified as wake-up call to galvanize action for mangrove restoration in other countries in the Southeast Asian region (Gaillard and Gomez, 2015). Despite this, mangroves are still being lost post-tsunami by deforestation to enable expansion of agriculture and aquaculture (Omar et al., 2019). Valuation of mangroves is likely to be useful if there is legal protection in place. However, the conflict between instituted federal policies and state-level policy implementation adds complexity to legal protection of mangroves (Amir, 2018).

Some recent national policy developments appear promising in terms of supportive governance that may improve uptake of ecosystem service valuation studies. The recently launched National Forestry Policy 2021 has streamlined what were previously three independent forestry policies by state (Peninsular Malaysia, Sabah and Sarawak). This revised policy places increased importance and focus on ecosystem services, particularly in relation to the implementation of mechanisms such as Payments for Forest Ecosystem Service and carbon emission reduction incentives. Moreover, the importance of cultural ecosystem services was recognized in the new forestry policy, which included strategies for promoting forestry-based ecotourism and preserving nature and indigenous heritage. Nevertheless, there is still a clear lack of cultural ecosystem services assessment as compared to provisional and regulating services in the Southeast Asian region (Hattam et al., 2021; Broszeit et al., 2022), indicating the need for future valuation studies to advance understanding of the cultural ecosystem services and their value in decision making.

On the other hand, the Central Bank of Malaysia is looking at understanding the risks associated with ecosystem services loss, with a view to incentivize protection of ecosystems *via* monetary practices aligned to sustainable national growth (Malaysia Bank Negara and World Bank, 2022). With the threat to natural resources now being more apparent, the demand for valuation and the likelihood of use of valuation results may be accelerated due to the urgency for action to protect or better manage natural resources (Waite et al., 2015). Therefore, valuation practitioners should carefully assess the current situation circumstances and tailor their valuation design to take full advantage of the enabled contextual conditions.

5 Conclusion

Despite the low number of documented successful applications of ecosystem service valuation in improved mangrove protection, valuation can play an important role in decision-making, when undertaken effectively and following best practices. This study identifies several recommendations for future ecosystem service valuation studies in Malaysia that can enable increased uptake of valuation outputs in support of sustainable mangrove management. The recommendations included strong, continual engagement with multi-stakeholder groups; the inclusion of stakeholder-driven scenarios that are relevant to the stakeholders in question; the adoption of standardised valuation units; and aligning valuation design and recommendation with existing national policies. The changing forest policy landscape within Malaysia provides a window of opportunity for enabling uptake of valuation findings. However, this requires a clear operationalization of ecosystem service concepts within decision making and policy development at all levels, as well as valuation practitioners well versed in valuation best practices.

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 authors.

Author contributions

SL: Investigation, Data curation, Writing - original draft, Writing - review & editing AT: Funding acquisition, Project administration, Writing - review & editing MA: Project administration, Writing - review & editing HG: Writing - review & editing CH: Funding acquisition, Writing - review & editing AE-J: Writing - review & editing. All authors contributed to the article and approved the submitted version.

Funding

This is a work from Nexus Action for Mangroves in Selangor, Malaysia: protection through value creation for community-public-private partnerships (NexAMS), a project under Universiti Malaya (grant no. IF025-2020) and University of Plymouth (grant no. NE/V005758/2), funded by the Newton Fund Impact Scheme (ID no. 537099481) and Malaysian Industry Government Group for High Technology (MIGHT).

Acknowledgments

The authors thank Sahadev Sharma for initial input in this work, and two reviewers for comments that improved the manuscript.

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.

Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2022.1033200/full#supplementary-material

References

Ahmad, S. (2009). Recreational values of mangrove forest in larut matang, perak. *J. Trop. For. Sci.* 21 (2), 81–87.

Aksnes, D. W., Langfeldt, L., and Wouters, P. (2019). Citations, citation indicators, and research quality: An overview of basic concepts and theories. *SAGE Open* 9 (1), 1–17. doi: 10.1177/2158244019829

Ament, L. A. (1994). Strategies for dissemination of policy research. *J. Nurse-Midwifery* 39 (5), 329–331. doi: 10.1016/0091-2182(94)90134-1

Amir, A. A. (2018). Mitigate risk for malaysia's mangroves. *Sci. March* 359, 1342–1343. doi: 10.1126/science.aas9139

Apostolopoulou, E., and Pantis, J. (2010). Development plans versus conservation: Explanation of emergent conflicts and state political handling. *Environ. Plann. A April* 42, 982–1000. doi: 10.1068/a42163

Arnautu, D., and Dagenais, C. (2021). Use and effectiveness of policy briefs as a knowledge transfer tool: a scoping review. *Humanities Soc. Sci. Commun.* 8, 211. doi: 10.1057/s41599-021-00885-9

Asma, W. I. W. T., Norhayati, M. T., and Lokman, M. H. (2012). The existing legislative, administrative and policy framework for the mangrove biodiversity management & conservation in Malaysia. *J. Politics Law* 5 (1), 180–188.

Avishek, K., Yu, X., and Liu, J. (2012). Ecosystem management in Asia pacific: Bridging science–policy gap. *Environ. Dev.* 3, 77–90. doi: 10.1016/j.envdev.2012.03.014

Aziz, A. A., Dargusch, P., Phinn, S., and Ward, A. (2015). Using REDD+ to balance timber production with conservation objectives in a mangrove forest in Malaysia. *Ecol. Economics* 120, 108–116. doi: 10.1016/j.ecolecon.2015.10.014

Bann, C. (1999). *A contingent valuation of the mangroves of benut, johor state, Malaysia* (Johor Bahru: Johor State Forestry Departmentt/DANCED/Darudec).

Barbier, E. B. (2007). Valuing ecosystem services for coastal wetland protection and restoration: Progress and challenges. *Resources* 2, 213–230. doi: 10.3390/resources2030213

Barbier, E., et al. (2011). The value of estuarine and coastal ecosystem services. *Ecol. Monogr.* Vol. 81 (2), 162–193.

Barnett, A., Fargione, J., and Smith, M. P. (2016). Mapping trade-offs in ecosystem services from reforestation in the Mississippi alluvial valley. *Bioscience* 66 (3), 223–237. doi: 10.1093/biosci/biv181

Bayraktarov, E., Saunders, M. I., Abdullah, S., Mills, M., Beher, J., Possingham, H. P., et al. (2016). The cost and feasibility of marine coastal restoration. *Ecol. Appl.* 26, 1055–1074. doi: 10.1890/15-1077

Beck, M. W., Heck, N., Narayan, S., Menéndez, P., Torres-Ortega, S., Losada, I. J., et al. (2020). *Reducing Caribbean risk: opportunities for cost-effective mangrove restoration and insurance* (Arlington, VA: The Nature Conservancy).

Bennet, E. L., and Reynolds, C. J. (1993). The value of a mangrove area in sarawak. *Biodivers. Conserv.* 2, 359–375. doi: 10.1007/BF00114040

Boyd, J., and Banzhaf, S. (2007). What are ecosystem services? the need for standardized environmental accounting units. *Ecol. Economics* 63, 616–626. doi: 10.1016/j.ecolecon.2007.01.002

Brander, L. M., Wagtendonk, A. J., Hussain, S. S., McVittie, A., Verburg, P. H., de Groot, R. S., et al. (2012). Ecosystem service values for mangroves in southeast Asia: A meta-analysis and value transfer application. *Ecosystem Serv.* 1, 62–69. doi: 10.1016/j.ecoser.2012.06.003

Broszeit, S., Hattam, C., Langmead, O., Praptiwi, R. A., Creencia, L., Then, A. Y.H., et al. (2022). *Ecosystem service provision by marine habitats in southeast Asia* (Plymouth, UK: PML Publishing).

Brown, K., Tompkins, E. L., and Adger, W. N. (2001). *Trade-off analysis for participatory coastal zone decision-making* (England: Overseas Development Group, University of East Anglia).

Bunting, P., Rosenqvist, A., Lucas, R. M., Rebelo, L. M., Hilarides, L., Thomas, N., et al. (2018). The global mangrove watch – a new 2010 global baseline of mangrove extent. *Remote Sens.* 10 (10), 1669. doi: 10.3390/rs10101669

Cagan, R. (2013). The San Francisco declaration on research assessment. *Dis. Models Mech.* 6, 869–870. doi: 10.1242/dmm.012955

Chew, L. L., Chong, V. C., Tanaka, K., and Sasekumar, A. (2012). Phytoplankton fuel the energy flow from zooplankton to small nekton in turbid mangrove waters. *Mar. Ecol. Prog. Ser.* 469, 7–14. doi: 10.3354/meps09997

Chong, V. C. (2005). "Fifteen years of fisheries research in the matang mangroves what have we learnt?," in *Sustainable management of matang mangroves: 100 years and beyond*. Ed. M. I. Shaharuddin, et al (Malaysia: Forest Biodiversity Series 4. Forestry Department Peninsular Malaysia.), 411–429.

Chong, V. C., Sasekumar, A., and Zgozi, S. W. (2012). "Ecology of fish and shrimp communities," in *IOES monograph series 14: Mangrove and coastal environment of selangor, Malaysia*. Eds. A. Sasekumar and V. C. Chong (Kuala Lumpur: University of Malaya Press), 215–241.

Cooper, E., Burke, L., and Bood, N. (2009). "Coastal capital: Belize," in *The economic contribution of belize's coral reefs and mangroves* (Washington DC: WRI Working Paper. World Resources Institute).

Costanza, R., de Groot, R., Sutton, P., van der Ploeg, S., Anderson, S. J., Kubiszewski, I., et al. (2014). Changes in the global value of ecosystem services. *Global Environ. Change* 26, 152–158. doi: 10.1016/j.gloenvcha.2014.04.002

Costanza, R., and Kubiszewski, I. (2012). The authorship structure of "ecosystem services" as a transdisciplinary field of scholarship. *Ecosystem Serv.* 1 (1), 16–25. doi: 10.1016/j.ecoser.2012.06.002

Daily, G., et al. (2009). Ecosystem services in decision making: Time to deliver. *Front. Ecol. Environ.* 7, 21–28. doi: 10.1890/080025

Damami, Z., MacKean, G., Bohm, E., DeMone, B., Wright, B., Noseworthy, T., et al. (2016). The use of a policy dialogue to facilitate evidence-informed policy development for improved access to care: the case of the Winnipeg central intake service (WCIS). *Health Res. Policy Syst.* 14, 78. doi: 10.1186/s12961-016-0149-5

Dang, A. N., Jackson, B. M., Benavidez, R., and Tomscha, S. A. (2021). Review of ecosystem service assessments: Pathways for policy integration in southeast Asia. *Ecosystem Serv.* 49, 101266. doi: 10.1016/j.ecoser.2021.101266

de Corte, K., Cairns, J., and Grieve, R. (2021). Stated versus revealed preferences: An approach to reduce bias. *Health Economics* 30, 1095–1123. doi: 10.1002/hec.4246

DEFRA (2007). *An introductory guide to valuing ecosystem services* (United Kingdom: Department for Environment, Food and Rural Affairs).

de Groot, R., Brander, L., van der Ploeg, S., Costanza, R., Florence, B., Braat, L., et al. (2012). Global estimates of the value of ecosystems and their services in monetary units. *Ecosystem Serv.* 1, 50–61. doi: 10.1016/j.ecoser.2012.07.005

de Groot, R. S., Stuip, M. A. M., Finlayson, C. M., and Davidson, N. (2006). "Valuing wetlands: Guidance for valuing the benefits derived from wetland ecosystem services," in *Ramsar technical report no. 3, gland* (Switzerland: Ramsar Convention Secretariat).

Duarte, C. M., Agusti, S., Barbier, E., Britten, G. L., Castilla, J. C., Gattuso, J. P., et al. (2020). Rebuilding marine life. *Nature* 580, 39–51. doi: 10.1038/s41586-020-2146-7

Ellison, A., Felson, A., and Friess, D. (2020). Mangrove rehabilitation and restoration as experimental adaptive management. *Front. Mar. Sci.* 7, 327. doi: 10.3389/fmars.2020.00327

Emerton, L. (2014). "Valuing & investing in ecosystems as development infrastructure: economic analysis of options for climate-compatible development in coastal zones of Kenya & Sri Lanka," in *iCOAST technical report* (Penicuik: LTS International Ltd).

Foong, S. Y., Yeap, C. A., and Butt, C. (2016). *A pilot rapid assessment of selected ecosystem services provided by the teluk air tawar-Kuala muda coast IBA in pulau pinang* (Kuala Lumpur: Malaysian Nature Society).

Friess, D., Thompson, B. S., Brown, B., Amir, A. A., Cameron, C., Koldewey, H. J., et al. (2016). Policy challenges and approaches for the conservation of mangrove forests in southeast Asia. *Conserv. Biol.* 30 (5), 933–949. doi: 10.1111/cobi.12784

Friess, D., Rogers, K., Lovelock, C. E., Krauss, K. W., Hamilton, S. E., Lee, S. Y., et al (2019). The state of the world's mangrove forests: Past, present, and future. *Annu. Rev. Environ. Resour.* 44, 1–27. doi: 10.1146/annurev-environ-101718-033302

Gaillard, J. C., and Gomez, C. (2015). Post-disaster research: Is there gold worth the rush? *Jamba: J. Disaster Risk Stud.* 7 (1), 1–6. doi: 10.4102/jamba.v7i1.120

Gilman, E. L., Ellison, J., Duke, N. C., and Field, C. (2008). Threats to mangroves from climate change and adaptation options: A review. *Aquat. Bot.* 89, 237–250. doi: 10.1016/j.aquabot.2007.12.009

Grimble, R. J., Aglionby, J., and Quan, J. (1994). "Tree resources and environmental policy: a stakeholder approach," in *NRI socio-economic series 7* (Chatham: Natural Resources Institute).

Haines-Young, R., and Potschin, M. B. (2018) *Common international classification of ecosystem services (CICES) V5.1 and guidance on the application of the revised structure* (Accessed 22 3 2022).

Haipeng, Z., and Xuxuan, X. (2012). Combining stated preference and revealed preference methods for the valuation of non-market goods*Chinese Journal of population resources and environment* 10, 121–126. doi: 10.1080/10042857.2012.10685119

Hamilton, S. E., and Casey, D. (2016). Creation of a high spatio-temporal resolution global database of continuous mangrove forest cover for the 21st century (CGMFC-21). *Global Ecol. Biogeogr.* 25, 729–738. doi: 10.1111/geb.12449

Hasan-Basri, B., Samdin, Z., and Ghani, A. N. (2020). Willingness to pay for conservation of mangrove forest in Kuala perlis, Malaysia. *Jurnal Ekonomi Malaysia* 54 (3), 89–99. doi: 10.17576/JEM-2020-5403-7

Hattam, C., Goh, H. C., Then, A. Y.H., Edwards-Jones, A., Ruslan, N. F.N., Yap, J. S.E., et al. (2020). Using nexus thinking to identify opportunities for mangrove management in the klang islands, Malaysia. *Estuarine Coast. Shelf Sci.* 247, 106917. doi: 10.1016/j.ecss.2020.106917

Hattam, C., Broszeit, S., Langmead, O., Praptiwi, R. A., Lim, V. C., Creencia, L. A., et al. (2021). A matrix approach to tropical marine ecosystem service assessments in south east Asia. *Ecosystem Serv.* 51, 101346. doi: 10.1016/j.ecoser.2021.101346

Hawkins, B., and Parkhurst, J. (2016). The 'good governance' of evidence in health policy. *Evidence Policy* 12 (4), 575–592.

Henrichs, T., Zurek, M., Eickhout, B., Kok, K., Raudsepp-Hearne, C., Ribeiro, T., et al. (2010). "Scenario development and analysis for forwards-looking ecosystem assessments," in *Ecosystems and human well-being: A manual for assessment practitioners*. Ed. N. Ash, et al (New York: Island Press), 151–220.

Himes-Cornell, A., Grose, S. O., and Pendleton, L. (2018a). Mangrove ecosystem service values and methodological approaches to valuation: Where do we stand? *Front. Mar. Sci.* 5, 376. doi: 10.3389/fmars.2018.00376

Himes-Cornell, A., Pendleton, L., and Atiyah, P. (2018b). Valuing ecosystem services from blue forests: A systematic review of the valuation of salt marshes, sea grass beds and mangrove forests. *Ecosystem Serv.* 30, 36–48. doi: 10.1016/j.ecoser.2018.01.006

Hong, L., Hemati, Z., and Zakaria, R. (2017). Carbon stock evaluation of selected mangrove forests in peninsular Malaysia and its potential market value. *J. Environ. Sci. Manage.* 20, 77–87. doi: 10.47125/jesam/2017_2/09

Jennings, V., Larson, L., and Yun, J. (2016). Advancing sustainability through urban green space: Cultural ecosystem services, equity, and social determinants of health. *Int. J. Environ. Res. Public Health* 13 (2), 196. doi: 10.3390/ijerph13020196

Jones, N., Jones, H., and Walsh, C. (2008). "Political science? strengthening science-policy dialogue in developing countries," in *Working paper*, vol. 294. (London: Overseas Development Institute).

Kaffashi, S., Radam, A., Shamsudin, M. N., Yacob, M., and Nordin, N. (2015). Ecological conservation, ecotourism, and sustainable management: The case of penang national park. *Forests* 6, 2345–2370. doi: 10.3390/f6072345

Kenter, J. O., O'Brien, L., Hockley, N., Ravenscroft, N., Fazey, I., Irvine, K. N., et al. (2015). What are shared and social values of ecosystems? *Ecol. Economics* 111, 86–99. doi: 10.1016/j.ecolecon.2015.01.006

Khuzaimah, Z., Ismail, M. H., and Mansor, S. (2013). "Mangrove changes analysis by remote sensing and evaluation of ecosystem service value in sungai merbok's mangrove forest reserve," in *Peninsular Malaysia* (Berlin, Heidelberg: Springer, Berlin, Heidelberg).

Kilpatrick, K., Carter, N., Bryant-Lukosius, D., Charbonneau-Smith, R., and DiCenso, A. (2015). The development of evidence briefs to transfer knowledge about advanced practice nursing roles to providers, policymakers and administrators. *Nurs. Leadership (Toronto Ont.)* 28 (1), 11–23. doi: 10.12927/cjnl.2015.24236

Lange, G.-M., and Jiddawi, N. (2009). Economic value of marine ecosystem services in Zanzibar: Implications for marine conservation and sustainable development. *Ocean Coast. Manage.* 52, 521–532. doi: 10.1016/j.ocecoaman.2009.08.005

Laurans, Y., Rankovic, A., Billé, R., Pirard, R., and Mermet, L. (2013). Use of ecosystem services economic valuation for decision making: Questioning a literature blindspot. *J. Environ. Manage.* 119, 208–219. doi: 10.1016/j.jenvman.2013.01.008

Leong, L. F., Kwan, K. H., Chong, V. C., and Sasekumar, A. (2005). "Resource valuation of Kuala selangor mangrove forest," in *Ecology of klang strait*. Eds. A. Sasekumar and V. C. Chong (Kuala Lumpur: University of Malaya Press), 237–259.

MacKinnon, A. J., Duinker, P. N., and Walker, T. R. (2018). *The application of science in environmental impact assessment* (England: Routledge Taylor & Francis Group).

Malaysia Bank Negara and World Bank (2022). *An exploration of nature-related financial risks in Malaysia* (Kuala Lumpur: International Bank for Reconstruction and Development/The World Bank).

Marre, J.-B., et al (2016). Is economic valuation of ecosystem services useful to decision-makers? lessons learned from Australian coastal and marine management. *J. Environ. Manage.* 178, 52–62. doi: 10.1016/j.jenvman.2016.04.014

Matthew, N. K., Shuib, A., Ramachandran, S., and Mohammad-Afandi, S. H. (2019). Economic valaution using travel cost method (TCM) in kilim karst geoforest park, langkawi, Malaysia. *J. Trop. For. Sci.* 31 (1), 78–89. doi: 10.26525/jtfs2019.31.1.078089

McKenzie, E., Irwin, F., Ranganathan, J., Hanson, C., Kousky, C., Bennett, K., et al. (2011). "Incorporating ecosystem services in decisions," in *Natural capital: Theory and practice of mapping ecosystem services*. Ed. P. Kareiva, et al (Oxford: Oxford University Press), 339–355.

McLeod, E., and Salm, R. V. (2006). *Managing mangroves for resilience to climate change* (Gland, Switzerland: IUCN).

McVittie, A., and Moran, D. (2010). Valuing the non-use benefits of marine conservation zones: An application to the UK marine bill. *Ecol. Economics* 70, 413–424. doi: 10.1016/j.ecolecon.2010.09.013

Mojiol, A. R., Kodoh, J., Wahab, R., Majuki, M., and Wahyudi, . (2016). Contribution of non-wood forest product to the local community living near mangrove forest in kudat, sabah. *J. Trop. Resour. Sustain. Sci.* 4, 38–41. doi: 10.36873/jht.v14i1.327

Mokthsim, N., and Osman Salleh, K. (2014). Malaysia's efforts toward achieving a sustainable development: Issues, challenges and prospects. *Proc. - Soc. Behav. Sci.* 120, 299–307. doi: 10.1016/j.sbspro.2014.02.107

Muhammad-Nor, S. M., Huxham, M., Salmona, Y., Duddy, S. J., Mazars-Simon, A., Mencuccinid, M., et al. (2019). Exceptionally high mangrove root production rates in the kelantan delta, malaysia; an experimental and comparative study. *For. Ecol. Manage.* 444, 214–224. doi: 10.1016/j.foreco.2019.04.026

Mukherjee, N., Sutherland, W. J., Dicks, L., Hugé, J., and Koedam and Dahdouh-Guebas, N. F. (2014). Ecosystem service valuations of mangrove ecosystems to inform decision making and future valuation exercises. *PLoS One* 9, 1–9. doi: 10.1371/journal.pone.0107706

Muralikrishna, I. V., and Manickam, V. (2017). "Environmental accounting," in *Environmental management: Science and engineering for industry*. Eds. I. V. Muralikrishna and V. Manickam (Oxford: Butterworth-Heinemann), 113–134.

Nabyonga-Orem, J., Gebrikidane, M., and Mwisongo, A. (2016). Assessing policy dialogues and the role of context: Liberian case study before and during the Ebola outbreak. *BMC Health Serv. Res.* 16 (Suppl. 4), 315–325. doi: 10.1186/s12913-016-1454-y

National Research Council (2005). *Valuing ecosystem services: Toward better* (Washington, DC: The National Academies Press).

Norhayati, A., Shukor, M. N., Juliana, S., and Wan Juliana, W. A. (2009). Mangrove flora and fauna of klang islands mangrove forest reserve, selangor, Malaysia. *Malaysian J. Sci.* 28 (3), 275–288. doi: 10.22452/mjs.vol28no3.6

Nursey-Bray, M. J., Vince, J., Scott, M., Haward, M., O'Toole, K., Smith, T., et al. (2014). Science into policy? discourse, coastal management and knowledge. *Environ. Sci. Policy* 38, 107–119. doi: 10.1016/j.envsci.2013.10.010

Olander, L., Johnston, R., Tallis, H., Kagan, J., Maquire, L., Boyd, J., et al. (2015). *Best practices for integrating ecosystem services into federal decision making* (Durham: National Ecosystem Services Partnership, Duke University).

Omar, H., Misman, M. A., and Musa, S. (2019). "GIS and remote sensing for mangroves mapping and monitoring," in *Geographic information system and science*. Eds. J. Rocha and P. Abrantes (London, UK: IntechOpen), 1–15.

Pendleton, L., Mongruel, R., Beaumont, N., Hooper, T., and Charles, M. (2015). A triage approach to improve the relevance of marine ecosystem services assessments. *Mar. Ecol. Prog. Ser.* 530, 183–193. doi: 10.3354/meps11111

Pourebrahim, S., Hadipour, M., and Mokhtar, M. B. (2011). Integration of spatial suitability analysis for land use planning in coastal areas; case of Kuala langat district, selangor, Malaysia. *Landscape Urban Plann.* 101, 84–97. doi: 10.1016/j.landurbplan.2011.01.007

Ramli, F., Sandin, Z., and Abdul Ghani, A. N. (2017). Willingness to pay for conservation fee using contingent valuation method: The case of matang mangrove forest reserve, perak, Malaysia. *Malaysian Forester* 80 (1), 99–110.

Rau, A.-L., Burkhardt, V., Dorninger, C., Hjort, C., Ibe, K., Keßler, L., et al. (2020). Temporal patterns in ecosystem services research: A review and three recommendations. *Ambio* 49 (8), 1377–1393. doi: 10.1007/s13280-019-01292-w

Raum, S. (2018). A framework for integrating systematic stakeholder analysis in ecosystem services research: Stakeholder mapping for forest ecosystem services in the UK. *Ecosystem Serv.* 29, 170–184. doi: 10.1016/j.ecoser.2018.01.001

Razak, N. A., Afandi, S. H. M., Shuib, A., and Ghani, A. N. A. (2018). Visitors travelling time cost for ecotourism at matang mangrove forest reserve. *Int. J. Business Soc.* 19 (1), 117–127.

Renard, Y. (2004). *Guidelines for stakeholder identification and analysis: A manual for Caribbean natural resource managers and planners* (Jamaica: Caribbean Natural Resources Institute).

Rivera-Planter, M., and Muñoz-Piña, C. (2005). Fees for reefs: economic instruments to protect mexico's marine natural areas. *Tourism* 8 (2-3), 195–213. doi: 10.1080/13683500508668214

Rogers, A. A., Kragt, M. E., Gibson, F. L., Burton, M. P., Petersen, E. H., Pannell, D. J., et al. (2015). Non-market valuation: usage and impacts in environmental policy and management in Australia. *Aust. J. Agric. Resource Economics* 59, 1–15. doi: 10.1111/1467-8489.12031

Rothstein, H. R., and Hopewell, S. (2009). "Grey literature," in *The handbook of research synthesis and meta-analysis*. Eds. H. Cooper, L. V. Hedges and J. C. Valentine (New York: Russell Sage Foundation), 103–125.

Ruslan, N. F. N., Goh, H. C., Hattam, C., Edwards-Jones, A., Moh, H. H., et al. (2022). Mangrove ecosystem services: Contribution to the well-being of the coastal communities in klang islands. *Mar. Policy* 144, 105222. doi: 10.1016/j.marpol.2022.105222

Salem, M. E., and Mercer, D. E. (2012). The economic value of mangroves: A meta-analysis. *Sustainability* 4, 359–383. doi: 10.3390/su4030359

Sasekumar, A., and Chong, V. C. (1991). *Fish and prawn communities in mangrove estuaries and mudflats in selangor (Malaysia)* (Manila: Marine Sciences Institute, University of Philippines).

Schneider, F., Giger, M., Harari, N., Moser, S., Oberlack, C., Providoli, I., et al. (2019). Transdisciplinary co-production of knowledge and sustainability transformations: Three generic mechanisms of impact generation. *Environ. Sci. Policy* 102, 26–35. doi: 10.1016/j.envsci.2019.08.017

Schuster, E., and Doerr, P. (2015). *A guide for incorporating ecosystem service* (Delmont, NJ: The Nature Conservancy, New Jersey Chapter).

Scoones, I. (2016). The politics of sustainability and development. *Annu. Rev. Environ. Resour.* 41, 293–319. doi: 10.1146/annurev-environ-110615-090039

Seglen, P. O. (1989). From bad to worse: Evaluation by journal impact. *Trends Biochem. Sci.* 14, 326–327. doi: 10.1016/0968-0004(89)90163-1

Seppelt, R., Fath, B., Burkhard, B., Fisher, J. L., Grêt-Regamey, A., Lautenbach, S., et al. (2012). Form follows function? proposing a blueprint for ecosystem service assessments based on reviews and case studies. *Ecol. Indic.* 21, 145–154. doi: 10.1016/j.ecolind.2011.09.003

Septiarani, B., and Handayani, W. (2016). The role of local champion in community-based adaptation in semarang coastal area. *Jurnal Pembangunan Wilayah Kota* 12 (3), 263–276. doi: 10.14710/pwk.v12i3.12901

Shahbudin, S., Zuhairi, A., and Kamaruzzaman, B. Y. (2012). Impact of coastal development on mangrove cover in kilim river, langkawi island, Malaysia. *J. Forestry Res.* 23 (2), 185–190. doi: 10.1007/s11676-011-0218-0

Shuib, A., Yii, A. B. S., and Edman, S. (2012). Conservation of deltaic mangrove forest resources in kuching, Sarawak: Local communities' willingness to pay. *Malaysian Forester* 75 (1), 65–72.

Stelk, M. J., and Christie, J. (2014). *Ecosystem service valuation for wetland restoration: What it is, how to do it, and best practice recommendations* (Windham, Maine: Association of State Wetland Managers).

Sunoto, Y. N., Fatiah, A. A., Ponrahono, Z., and Osman, M. M. (2020). Profiling the perceived mangrove forest use value and community's willingness to pay for mangrove conservation. *Plann. Malaysia: J. Malaysian Institute Planners* 18 (3), 229–240. doi: 10.21837/pm.v18i13.788

Suter, E., and Armitage, G. D. (2011). Use of a knowledge synthesis by decision makers and planners to facilitate system level integration in a large Canadian provincial health authority. *Int. J. Integrated Care* 11, e011. doi: 10.5334/ijic.576

TEEB (2010). *The economics of ecosystems and biodiversity ecological and economic foundations* (London and Washington: Earthscan).

Then, A. Y. H., and Chong, V. C. (2022). Trophic guild structure and dietary patterns of a juvenile-dominated demersal fish community in a tropical mangrove estuarine system. *Bull. Mar. Sci* 98, 271–296. doi: 10.5343/bms.2022.0001

Tisdell, C. A., and Xue, D. (2013). "Managing ecosystem services for human benefit: Economic and environmental policy challenges," in *Environmental policy: Management, legal issues and health aspects*. Eds. E. Crighton and P. Davovich (New York: Nova Science Publishers), 87–106.

Torres, C., and Hanley, N. (2017). Communicating research on the economic valuation of coastal and marine ecosystem services. *Mar. Policy* 75, 99–107. doi: 10.1016/j.marpol.2016.10.017

Tranggono, D., Nuryananda, P. F., and Putra, A. Y. T. (2021). Local champion: Communication characteristics in community empowerment based on local innovation. *J. Students Acad. Res.* 7, 72–84. doi: 10.35457/josar.v7i1.1535

Uzochukwu, B., Onwujekwe, O., Mbachu, C., Okwuosa, C., Etiaba, E., Nyström, M., et al. (2016). The challenge of bridging the gap between researchers and policy makers: Experiences of a health policy research group in engaging policy makers to support evidence informed policy making in Nigeria. *Globalization Health* 12, 67. doi: 10.1186/s12992-016-0209-1

van Oudenhoven, A. P. E., Siahainenia, A. J., Sualia, I., Tonneijck, F. H., van der Ploeg, S., de Groot, R. S., et al. (2015). Effects of different management regimes on mangrove ecosystem services in Java, Indonesia. *Ocean Coast. Manage.* 116, 353–367. doi: 10.1016/j.ocecoaman.2015.08.003

Vatn, A., and Bromley, D. W. (1994). Choices without prices without apologies. *J. Environ. Economics Manage.* 26, 129–148. doi: 10.1006/jeem.1994.1008

Wainger, L., and Mazzotta, M. (2011). Realizing the potential of ecosystem services: A framework for relating ecological changes to economic benefits. *Environ. Manage.* 48, 710. doi: 10.1007/s00267-011-9726-0

Waite, R., Kushner, B., Jungwiwattanaporn, M., Gray, E., and Burke, L. (2015). Use of coastal economic valuation in decision making in the Caribbean: Enabling conditions and lessons learned. *Ecosystem Serv.* 11, 45–55. doi: 10.1016/j.ecoser.2014.07.010

Wiesmann, U., Liechti, K., and Rist, S. (2005). Between conservation and development: Concretizing the first world natural heritage site in the Alps through participatory processes. *Mountain Res. Dev.* 25, 128–138. doi: 10.1659/0276-4741(2005)025[0128:BCAD]2.0.CO;2

Yakob, M. R., Radam, A., and Shuib, A. (2009). A contingent valuation study of marine parks ecotourism: The case of pulau payar and pulau redang in Malaysia. *J. Sustain. Dev.* 2 (2), 95–105. doi: 10.5539/jsd.v2n2p95

Young, J. C., Waylen, K. A., Sarkki, S., Albon, S., Baibridge, I., Balian, E., et al. (2014). Improving the science-policy dialogue to meet the challenges of biodiversity conservation: having conversations rather than talking at one-another. *Biodiver. Conserv.* 23 (2), 387–404. doi: 10.1007/s10531-013-0607-0

Zelenski, J. M., Dopko, R. L., and Capaldi, C. A. (2015). Cooperation is in our nature: Nature exposure may promote cooperative and environmentally sustainable behavior. *J. Environ. Psychol.* 42, 24–31. doi: 10.1016/j.jenvp.2015.01.005

Zeng, Y., Friess, D. A., Sarira, T. V., Siman, K., and Koh, L. P. (2021). Global potential and limits of mangrove blue carbon for climate change mitigation. *Curr. Biol.* 31 (8), 1737–1743. doi: 10.1016/j.cub.2021.01.070

frontiers | Frontiers in Marine Science

TYPE Original Research
PUBLISHED 01 December 2022
DOI 10.3389/fmars.2022.1043943

Check for updates

OPEN ACCESS

EDITED BY
Lorenzo Mari,
Politecnico di Milano, Italy

REVIEWED BY
Huong Nguyen,
Murdoch University, Australia
Michael Alister Reid,
University of New England, Australia

*CORRESPONDENCE
Pham Hong Tinh
phtinh@hunre.edu.vn

SPECIALTY SECTION
This article was submitted to Marine Conservation and Sustainability, a section of the journal Frontiers in Marine Science

RECEIVED 14 September 2022
ACCEPTED 14 November 2022
PUBLISHED 01 December 2022

CITATION
Tinh PH, MacKenzie RA, Hung TD, Vinh TV, Ha HT, Lam MH, Hanh NTH, Tung NX, Hai PM and Huyen BT (2022) Mangrove restoration in Vietnamese Mekong Delta during 2015-2020: Achievements and challenges. *Front. Mar. Sci.* 9:1043943. doi: 10.3389/fmars.2022.1043943

Mangrove restoration in Vietnamese Mekong Delta during 2015-2020: Achievements and challenges

Pham Hong Tinh[1*], Richard A. MacKenzie[2], Tran Dang Hung[3], Truong Van Vinh[4], Hoang Thi Ha[1], Mai Huong Lam[1], Nguyen Thi Hong Hanh[1], Nguyen Xuan Tung[5], Pham Minh Hai[6] and Bui Thanh Huyen[7]

[1]Faculty of Environment, Hanoi University of Natural Resources and Environment, Hanoi, Vietnam, [2]Pacific Southwest Research Station, United States Department of Agriculture (USDA) Forest Service, Institute of Pacific Islands Forestry, Hilo, HI, United States, [3]Vietnam Institute of Meteorology, Hydrology and Climate Change, Hanoi, Vietnam, [4]Faculty of Forestry, Nong Lam University, Ho Chi Minh, Vietnam, [5]Mangrove Ecosystem Research Center, Hanoi National University of Education, Hanoi, Vietnam, [6]Vietnam Institute of Geodesy and Cartography, Ministry of Natural Resources and Environment, Hanoi, Vietnam, [7]Department of Water-Environment-Oceanography, University of Science and Technology of Hanoi, Hanoi, Vietnam

Mangrove forest in the Mekong Delta plays important roles in protecting coasts from soil erosion and strong waves, supplying seafood, and accumulating carbon. Despite these benefits, mangroves have been and continue to be severely damaged by the impacts of natural and socioeconomic activities. In recent years, large areas of mangrove forest have been restored through planting and other various management actions. In this study, we analyzed high-resolution WorldView-2 images to quantify changes in the mangrove forest in seven coastal provinces (Tien Giang, Ben Tre, Tra Vinh, Soc Trang, Bac Lieu, Ca Mau, and Kien Giang) of the Mekong Delta from 2015 to 2020. Our study is one of the first to analyze mangrove forest change at the commune scale, the smallest official administrative unit in Vietnam, to determine the area of restored mangroves. The potentials and challenges in future mangrove restoration were also assessed by analyzing satellite imagery and field survey data. In the study area, mangrove forest area increased by 11,184 ha (approximately 2,237 ha per year) from 79,593 ha in 2015 to 90,777 ha in 2020. A total area of 16,138 ha (approximately 20.3%) was lost due to mangrove conversion to other land uses, aquaculture activities and coastal erosion, etc., while 27,322 ha (approximately 34.1%) was restored or newly planted during state- and NGO-funded mangrove restoration projects and programs. These results confirmed that mangrove restoration projects and programs have played a significant role in maintaining and increasing mangrove forest cover in Mekong Delta. The results can also assist managers and decision makers in mangrove restoration evaluation, and suggest analyzing WorldView-2 images to monitor mangrove restoration over time in Vietnam.

KEYWORDS

mangroves, restoration, worldview, Mekong, Vietnam

Introduction

Vietnam used to be known as one of the countries with a large area of mangroves in the world. In 1943, 408,500 ha of mangroves were distributed along its coastline, from Quang Ninh province in the north to Kien Giang province in the south (Hong and San, 1993; Tinh et al., 2022). These mangrove forests providing valuable ecosystem services to human populations that include: 1) coastal protection from storms, floods, and erosion, 2) carbon sequestration for climate change adaptation and mitigation, 3) the provision of fuel and raw material products, 4) habitat for aquatic and terrestrial species, and 5) and other social, human, financial and physical capital for local livelihoods (Hawkins et al., 2010; Pham et al., 2022). However, Vietnam mangrove forests were severely damaged by the Second Indochina War from 1955-1975 (Hong and San, 1993), the shrimp aquaculture boom of the 1980s and 1990s (de Graaf and Xuan, 1998), and impacts from climate change (e.g., sea level rise, increased storms) in recent decades (Ward et al., 2016; Truong et al., 2017).

The Mekong Delta possesses around 84% of Vietnam's mangroves (Tinh et al., 2022). According to the Circular No. 34/2009/TT-BNNPTNT of the Ministry of Agriculture and Rural Development (MARD), Mekong Delta's mangroves were classified into protection forest (protect soil and water resources, prevent erosion and desertification, limit disasters, regulate climate and protect the environment), special-use forest (natural preservation of national ecological standard samples and forest biological gene sources; scientific research; protection of historical and cultural relics and places of scenic beauty) and production forest (production and trading of timber, non-timber forest products). Due to the natural impacts and socioeconomic activities described above, mangrove area in Mekong region has significantly declined (de Graaf and Xuan, 1998; Hong et al., 2019; Liu et al., 2020). Phan and Stive (2022) reported that the total area of mangroves in Mekong Delta decreased from 185,800 ha in 1973 to 102,160 ha in 2020, with a loss of 2,150 ha per year to aquaculture expansion and 430 ha per year to coastal erosion. As a result, various policies and projects on mangrove restoration, rehabilitation, and plantations have been implemented in the Mekong Delta, including a state project on the protection and development of coastal forests during 2015-2020 and other mangrove projects funded by the World Bank, Oxfam, etc. (Pham et al., 2022).

Previous studies revealed that the condition and area of mangrove forest in the Mekong Delta was deteriorating and shrinking, respectively (Tong et al., 2004; Binh et al., 2005; Thu and Populus, 2007; Quyen, 2011; Bullock et al., 2017; Truong and Do, 2018; Hong et al., 2019; Liu et al., 2020; Phan and Stive, 2022; Pham et al., 2022). Tinh et al. (2022), however, reported a net gain in mangrove forest area through restoration/reforestation efforts by the Vietnamese government as well as other national and international organizations. Most of these remote sensing studies used Landsat images with a resolution 30 m (Bullock et al., 2017; Hong et al., 2019; Liu et al., 2020; Phan and Stive, 2022; Tinh et al., 2022) or Sentinel images with a resolution of 10 m (Pham et al., 2022). Only a few of these studies used higher resolution images (SPOT) and these studies only focused on one or two provinces (Tong et al., 2004; Thu and Populus, 2007). Because mangroves in Mekong Delta often grow along narrow areas of coastline or are fragmented into small patches, detailed mangrove ecosystem characterization becomes difficult with moderate-resolution satellite data (Green et al., 1998). Therefore, this study used high-resolution (1.84 m) WorldView-2 imagery to quantify changes in the mangrove forest along the Mekong Delta coast from 2015 to 2020. We analyzed the mangrove forest change at commune scale to determine the areas where mangroves were restored or lost. The challenges in mangrove restoration was also discussed. The results from this study will help Vietnam and the coastal provinces of the Mekong Delta assess mangrove restoration efforts during the past period 2015-2020 as well as the next periods from 2021-2025 and 2021-2030. This latter period is particularly important as a project on the protection and development of coastal forests from 2021- 2030 was recently approved by the Vietnam Government on 10 October 2021 (MARD, 2021; Pham et al., 2022).

Materials and methods

Study area

This study was carried out in seven provinces (Tien Giang, Ben Tre, Tra Vinh, Soc Trang, Bac Lieu, Ca Mau, Kien Giang) across the coastal area of the Mekong Delta, including 99 coastal communes within 26 different districts (Figure 1). This tropical region is characterized by a short dry season from January to March and a more extended rainy season from April to December (Tong et al., 2004), an average temperature of 25-27°C, and an annual precipitation of 1,600-2,000 mm (Nguyen and Nguyen, 2013). The Mekong Delta has a low-lying topography, receives abundant nutrient-rich alluvial deposits from the Mekong and Dong Nai rivers (Veettil et al., 2019), and contains the largest area of Vietnam's mangrove forest with a total of 69 mangrove species (Hong and San, 1993). The mangrove forests in the Mekong Delta are typically dominated by *Rhizophora apiculata*, *R. mucronata*, *Avicennia alba*, *A. officinalis*, *Sonneratia alba*, *Bruguiera cylindrixa*, *B. parviflora* (Tri, 1999). Numerous mangrove restoration efforts were carried out between 2015 and 2020 in response to the large areas of mangroves that have been deforested and degraded from mangrove conversion to other land uses and the effects of climate change (Pham et al., 2022).

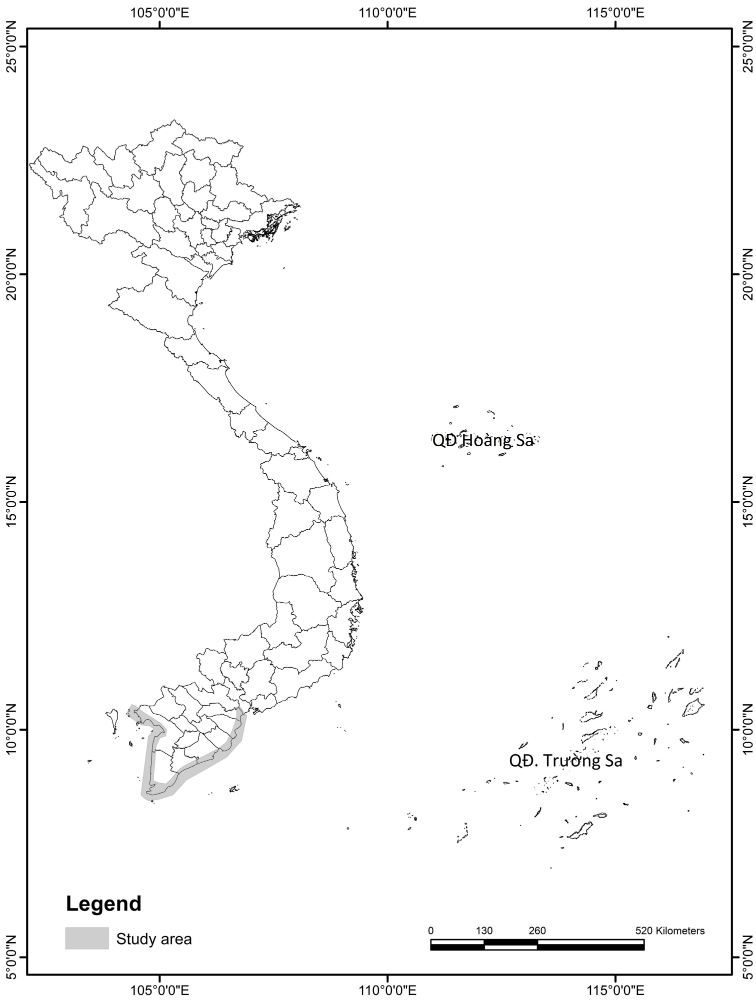


FIGURE 1
Study area in Mekong Delta, Vietnam.

Satellite data

In this study, both WorldView-2 and Sentinel-2 (Sentinel-2A and 2B) images were used. WorldView-2 is a high-resolution satellite that was launched on October 8, 2009 to collect panchromatic imagery of 0.46 m resolution, and eight-band multispectral imagery with 1.84 m resolution. These represent some of the highest resolution satellite images available. A total of 118 WV-2 images covering the study area (56 images for 2015 and 82 images for 2020) were obtained from the Polar Geospatial Center (University of Minnesota) to track changes in mangrove extent. The average cloud cover of the WV-2 images over the study area for 2015 and 2020 were 16.38% and 23.94%, respectively. Sentinal-2A was launched on June 23, 2015 and Sentinel-2B was launched on March 7, 2017 to acquire optical imagery with 13 bands that consist of 4 bands with 10 meters spatial resolution, 6 bands with 20 meters, and 3 bands with 60 meters. In this study, eight Sentinel-2 images with an average cloud cover over of less than 5% were downloaded from the Copernicus Open Access Hub (https://scihub.copernicus.eu/dhus/#/home), georeferenced to 1.84 m resolution to ensure proper alignment, and compensated for frequent cloud cover of WV-2 images (Alm et al., 2020). The Sentinel-2 images were georeferenced to 1.84 m WV-2 imagery to ensure proper alignment. As both WV-2 and Sentinel-2 images used in this study were single-date imagery, we only selected images taken near or at low tide to minimize the amount of mangrove area that was submerged by the tide.

Ground truth data

Ground truth data was collected from June 05 to 19, 2022, to describe land cover at 250 randomly selected reference points that contained either mangrove forest (150 points) or other land cover (100 points) for proper imagery classification. A Garmin GPSMAP64 with approximately 3 m accuracy was used to take GPS waypoints at each visited ground reference point. Photographs were also taken at each point to ensure that the area at each point was correctly described. Ground reference points were later used to calibrate and assess the accuracy of the land use classification. In order to better understand the current management status and use of mangrove forest, we also conducted semi-structured interviews in six communes (Binh An, Thuan Hoa, Van Khanh Dong, Vien An, Dat Moi and Tan An) with the highest percentage of mangrove loss. A total of 100 people (12 forest rangers, 8 local guards, 10 local authorities from the Commune People's Committee, and 70 local villagers living nearby and under mangrove canopy) were interviewed during the ground truth survey. The semi-structured interview (Supplementary Table S1) contained a series of questions that provided us with qualitative data on past and present mangrove management and restoration, natural and social-economic activities affecting mangrove, and historical mangrove distribution change. This interview-based data was used in combination with remote sensing-based estimate and published information to discuss the successes and challenges of mangrove restoration in the Mekong Delta.

Satellite imagery processing

The WV-2 images for 2015 and 2020 were pre-processed using the FLAASH (Fast Line-of-sight Atmospheric Analysis of Spectral Hypercubes) algorithm in ENVI 5.3 to correct the atmosphere. As cloud cover is recognized as a significant loss of data and information quality, the cloud portions of WV-2 images were detected and removed using the Con tool in ArcGIS Desktop. These gaps were then filled with the cloud free Sentinel-2 images taken during the same year (Das et al., 2020). These compensated images were then clipped to extract the areas of interest where mangroves were more likely to be present (e.g., low-lying areas, intertidal zones and estuaries) for further analysis. Images were then segmented into homogeneous objects using eCognition software application and classified into two classes: mangroves and non-mangroves using the supervised image classification technique of Maximum Likelihood Classification (Islam et al., 2019; Thakur et al., 2021). Non-mangrove land class included agricultural, commercial and industrial area, residential area, roads, exposed soils, bare land and open water. In the image classification, 500 training samples (250 for each class) were randomly selected using Google Earth and existing land use maps.

The validation data collected from ground truthing land cover maps and Google Earth for the year 2020 and from land cover maps and Google Earth for the year 2015 was used to assess land cover classification accuracy. The validation points of each land cover class were converted to raster. Validation raster was then snapped to each land cover layer and compared to calculate metrics of accuracy that include producer's accuracy, user's accuracy, overall accuracy, and overall kappa statistic (Landis and Koch, 1977; Fisher et al., 2018).

Results

Mangrove classification and spatial distribution showed that from 2015 to 2020, Mekong Delta mangroves were distributed along the study area, with the greatest concentration of area in Ca Mau province and narrow coastal strips in other provinces (Figure 2). The overall accuracy of the classified maps for 2015 and 2020 were 90.33% and 93.00%, respectively, and the kappa coefficient were 0.81 and 0.86, respectively (Table 1). These accuracy metrics showed an acceptable agreement between the classification results and reference data (Dan et al., 2016; Thomas et al., 2018; Nguyen et al., 2020b).

Due to the influence of aquaculture under the forest canopy over the past several decades and the dramatically growing erosion of the shoreline in recent years, mangrove forests in the western coastal area were much more fragmented than along the eastern coast. Mangroves were more unified and well-developed on the eastern and southern coastal area. In Ca Mau Cape where mangroves were well-managed by national parks and protection forest management boards, mangrove forests were dense and homogeneous. A large area of mangrove forest designated as production forest, was found lying further inland, which was given to the local villagers for reforestation and alternate harvesting once the trees reached the proper size.

In the study area, mangrove forest covered 79,593 ha in 2015 and increased to 90,777 ha in 2020. The loss of mangroves from 2015 to 2020 was 16,138 ha (approximately 20.3%), while 27,322 ha (approximately 34.1%) was restored or newly planted. As a consequence, the Mekong Delta's overall net area of mangrove forest increased by 11,184 ha (approximately 2,237 ha per year). Classification findings indicated that almost 80% of mangrove area remained unchanged. A more detailed analysis at the commune level revealed variations in the rate of mangrove change. Out of the 102 communes in the study area, mangrove area increased in 54 communes but decreased in the remaining 48 communes (Figure 3). The communes with mangrove area loss were mainly in the western Kien Giang province and in the eastern Tra Vinh and Bac Lieu provinces while the provinces of Ca Mau, Ben Tre, and Soc Trang had the majority of communes that experienced increasing mangrove areas. The communes with the most significant increase in mangrove area were Vien An Dong (2,810 ha), Lam Hai (1,860 ha), Vien An (1,894 ha), Rach Goc (1,020 ha), Tan An Tay (993 ha), and Dat Mui (748 ha). In contrast, Vinh Hai (-185 ha), Binh An (-160 ha), Vinh Thinh (-145 ha), Tan Thuan (-86 ha), and Thuan Hoa (-83 ha) communes lost the largest areas of mangroves (Figure 3; Supplementary Table S2). Supplementary Table S2 also

10.3389/fmars.2022.1043943

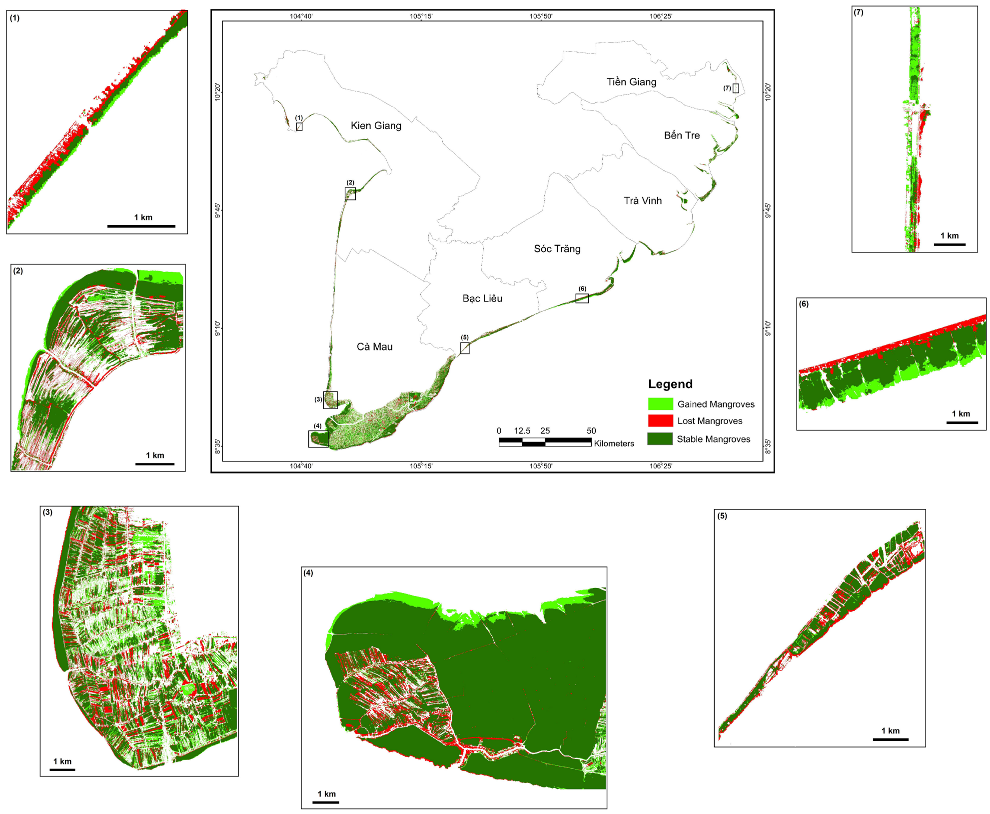


FIGURE 2
Spatial distribution of existing (stable) mangroves and mangrove change in the Mekong Delta from 2015-2020

provides more detailed information on the area of mangroves gained and lost for the entire study area.

Discussion

Achievements in mangrove restoration

Both natural and anthropogenic factors have caused major losses in Vietnam's mangrove area and distribution (Hawkins et al., 2010; Dat and Yoshino, 2011; Pham and Yoshino, 2016; Hauser et al., 2017; Nguyen et al., 2019). In this study, analyses of WorldView-2 data for the period of 2015 to 2020 revealed that mangrove area has increased by 11,184 ha in seven provinces of the Mekong Delta (approximately 2.8% increase by area per year) (Supplementary Table S2). The rate of mangrove increase in this study area is higher than the estimate of 0.4% by area per year for nine southern provinces (include Ho Chi Minh city, Ba Ria - Vung Tau and 07 provinces of the Mekong Delta) (Hawkins et al., 2010; Pham et al., 2022), lower than in other parts of the country like Thanh Hoa in the north where mangroves increased up to 16% by area per year (Nguyen et al., 2020a) and much better than many other parts of the world where mangrove loss is still occurring (Toosi et al., 2019; Halder et al., 2021; Kiprono,

TABLE 1 Accuracy metrics for land cover classification.

Land use classes	2020		2015	
	Producer's accuracy (%)	User's accuracy (%)	Producer's accuracy (%)	User's accuracy (%)
Mangroves	91.61	94.67	88.54	92.67
Other	94.48	91.33	92.31	88.00
Overall accuracy	93.00		90.33	
Kappa coefficient	0.86		0.81	

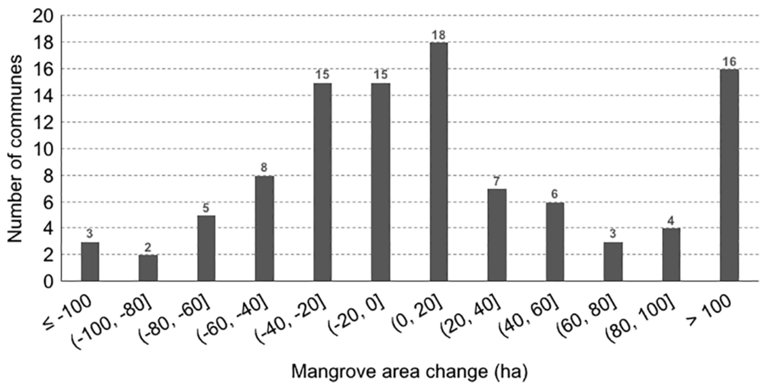


FIGURE 3
Histogram showing the number of communes that lost and gained mangroves during 2015-2020.

2021). Beside the differences in time periods and the extent of study areas, the resolution of the satellite images used also caused the differences in estimates of mangrove change. Vietnam's mangrove forests are mostly distributed in elongated or fragmented patches which are often narrower or smaller than the pixel size of medium-resolution imagery with 10-100 m resolution. Xia et al. (2018) discussed mapping mangrove forests with high-resolution imagery with several meters resolution can produce results with the highest accuracy to date.

The mangrove area increase is likely due to the long history of successful mangrove restoration efforts that have occurred in Vietnam (Hai et al., 2020). Vietnam has a strong legal framework that emphasizes the need to enhance mangrove restoration and the financial commitments from government to fund these efforts. Each province in the study area has also issued policies on mangrove management and restoration (Pham et al., 2022). This has resulted in many restoration programs and projects that have been implemented since the early 1990s (Hai et al., 2020). Furthermore, national projects like the National Target Program to Respond to Climate Change and Green Growth from 2016-2020 (Decision No.1670/QD-TTg dated 31 October 2017) and other related programs resulted in 39 projects that were implemented to protect and develop coastal forests in seven coastal provinces of the study area (Table 2). In addition to these state-funded projects, mangrove restoration in Mekong Delta has also received support from international NGOs that include KFW Development Bank (MARD, 2014), Green Climate Fund (SNV, 2016), The United Nations Development Program (UNDP, 2015), International Climate Initiative (MARD, 2014) and World Bank (WB, 2017). These projects and initiatives all contributed to successful mangrove restoration projects and programs that have played a significant role in maintaining and increasing mangrove forest cover in Mekong Delta (Figure 4).

Mangroves are one of the most carbon-rich ecosystems in tropical regions (Alongi, 2012; Donato et al., 2012). In the Mekong Delta, Nam et al. (2016) reported that carbon stocks from natural mangroves and 35 year old restored mangroves were not different. This suggests that restoration can quickly return C to degraded/deforested mangroves and that mangrove restoration plays a significant role in carbon emission reduction strategies and

TABLE 2 Mangrove restoration projects in Mekong Delta during 2015-2020.

No.	Province	Total number of projects	Mangrove project area		
			New plantation	Restoration	Protection
1	Tien Giang	4	150	–	–
2	Ben Tre	4	221	–	4,236
3	Tra Vinh	11	695	–	10,185
4	Soc Trang	6	1,864	850	23,426
5	Bac Lieu	3	208	44	–
6	Ca Mau	7	1,330	1,162	49,000
7	Kien Giang	4	832	–	1,331
Total		39	5,300	2,056	88,178

Sources: MARD (2021) and Pham et al. (2022).

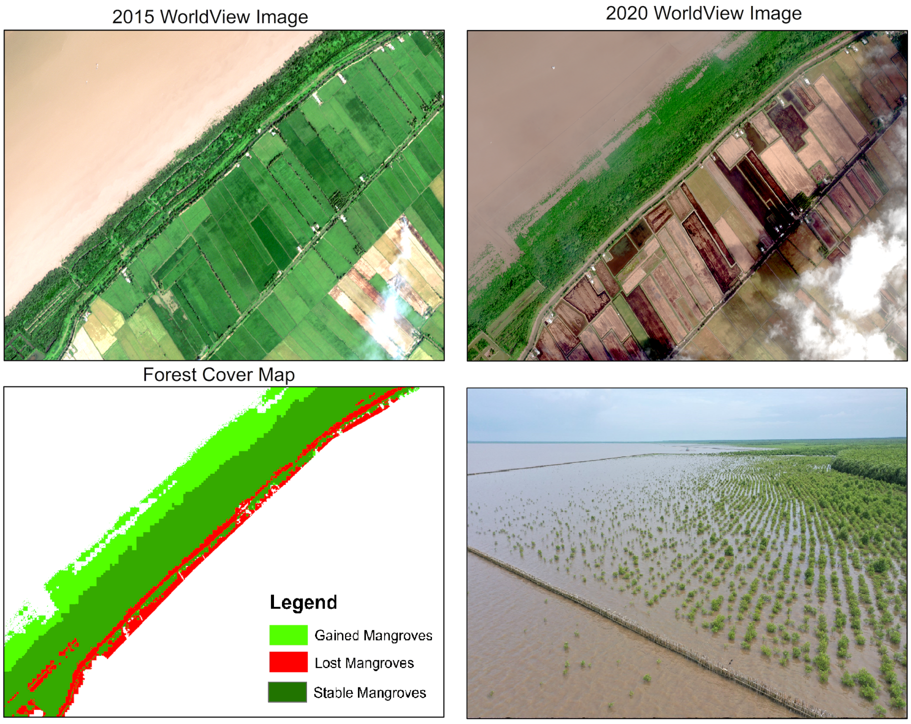


FIGURE 4
Mangroves are successfully restored to the sea and attributed to high sediment deposition and land creation that occurs along these shorelines.

should be integrated into related international agreements. In Vietnam's Intended Nationally Determined Contributions, mangrove plantations were proposed as one of the important options for climate change mitigation (Hai et al., 2020) and could help Vietnam will reach its net-zero carbon emission target by 2050. Vietnam has also committed to reducing greenhouse gas emissions by 9% with domestic resources and 27% with international support by 2030. Moreover, with the adoption of the Paris Agreement, 29 Parties that include Vietnam have committed to using mangrove restoration as a climate mitigation activity (Hai et al., 2020). These could be important basis for the boom of mangrove restoration projects in the coming years, especially when restoration projects are well designed. For the 2021-2025 period, the Mekong Delta provinces have designated large areas (approximately 91,000 ha) for mangrove restoration and afforestation. These areas have been allocated for forestry purposes but are either currently without forest cover or have newly planted forests with low survival rates that could be enriched with additional planting (MARD, 2021; Pham et al., 2022).

Challenges in mangrove restoration

While mangrove area has increased in the entire study area, 48 of 99 study communes have experienced mangrove loss. Increases in mangrove forest area are concentrated in only a few communes (Figure 2; Supplementary Table S2) and mostly in production forest areas where trees will be harvested as soon as they reach a proper size. This means mangroves still remain at high risk of deforestation. Furthermore, while a large area has been designated for mangrove restoration, those restoration projects continue to face many challenges from both natural and anthropogenic impacts. First, mangrove forests are usually distributed in narrow strips along the coast that are vulnerable to coastal squeeze that results from erosion at the oceanic mangrove interface and development at the mangrove upland interface that limits the ability of mangrove to migrate inland (Phan et al., 2015; Truong et al., 2017). Coastline erosion and the construction of sea dikes to create more inland space for fish/shrimp farming and cultivation as well as to prevent salinity intrusion were both observed at many of our sampled sites (Figure 5). Second, the conversion of mangroves to other land uses that are still active and that include agriculture, aquaculture and other socioeconomic activities (Figure 6) in areas planned for mangrove restoration limits the success and effectiveness as well as justification for restoring those areas (Pham et al., 2022). Third, management related issues are also likely to have a decisive impact on mangrove restoration in Mekong Delta (Hai et al., 2020). Co-management was an effective way of maintaining and enhancing the protection function of the mangrove forest while at the same

10.3389/fmars.2022.1043943

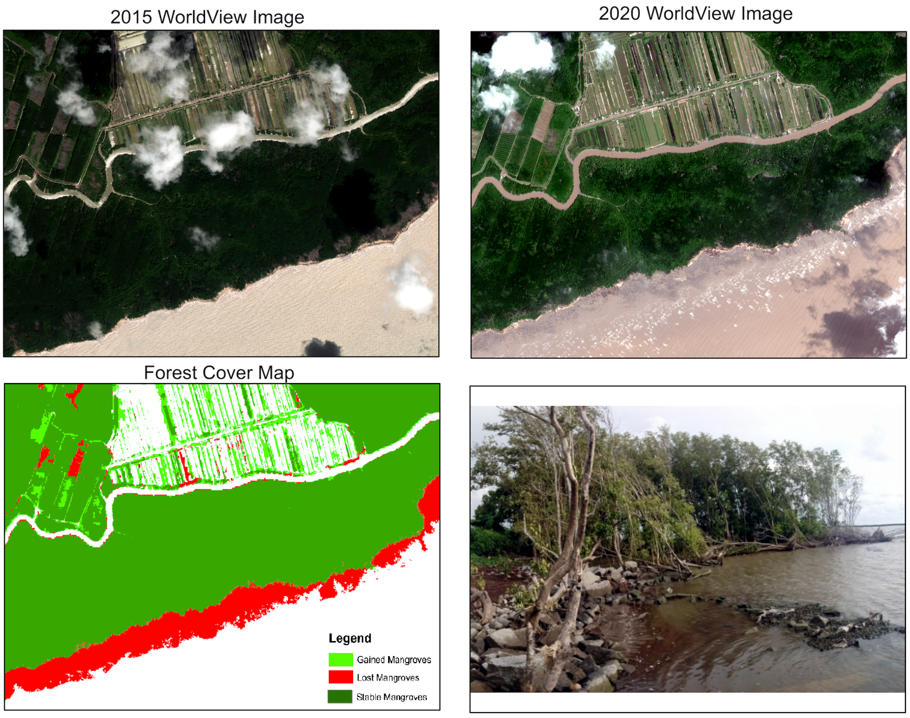


FIGURE 5
Mangrove squeeze that results from coastal erosion and inland barriers that prevent mangrove migration.

FIGURE 6
Mangrove conversion to other land uses (road and intensive aquaculture).

time providing livelihood for local communities. The integrated mangrove aquaculture systems such as mud crab fattening in mangrove pens and cages, mixed shrimp-mangrove-crab-cockle systems or integrated mangrove fish or shrimp farms were considered an effective solution to improve local livelihoods (Macintosh and Ashton, 2003). However, such systems also generated a potential challenge for mangrove restoration and management as the farmers illegally cut down mangroves or gradually cut down the roots of mangroves to weaken or kill trees in order to increase their aquaculture production (Figure 7). To solve this issue, a pilot project managed by the Forest Ranger of Bien Tay Protection Forest in 2015 was carried out to resettle the villagers, who implemented integrated mangrove aquaculture systems in Khanh Hoi, Khanh Binh Tay, Khanh Hai and Song Doc communes to inland areas and mangroves will be restored within their ponds. However, until now, hundreds of households have yet to move to a new place due to the lack of a new livelihood to support them.

The results of this study showed that the mangrove area of Mekong Delta increased by 11,184 ha during the 2015-2020 period (approximately 2,237 ha per year). The mangrove area increased in 54 communes that were mainly in eastern and southern provinces (e.g. Ca Mau, Ben Tre, and Soc Trang), but declined in the remaining 48 communes that were mainly in the western Kien Giang province. The state-funded mangrove restoration projects under the National Target Program to Respond to Climate Change and Green Growth for 2016-2020 and related programs, and other mangrove restoration projects supported from the international NGOs have played a significant role in maintaining and increasing mangrove forest cover in Mekong Delta. For the 2021-2025 and 2026-2030 periods, a large area has been designated for mangrove restoration under the National Program on Protection and Development of Forests in Coastal Areas to Respond to Climate Change and Promote Green Growth for 2021-2030. These mangrove restoration projects, however, will face challenges from both natural and anthropogenic impacts (e.g. coastal erosion, mangrove conversion and aquaculture activities). The results from this study could help Vietnam assess successful and failed mangrove restoration results from 2015-2020 in order to identify factors that will increase future restoration project that will occur from 2021-2025 and 2026-2030, when new state programs on the protection and development of coastal forests will be implemented. Results could also be used to identify degraded mangrove areas that could be prioritized for restoration as it is within these areas where

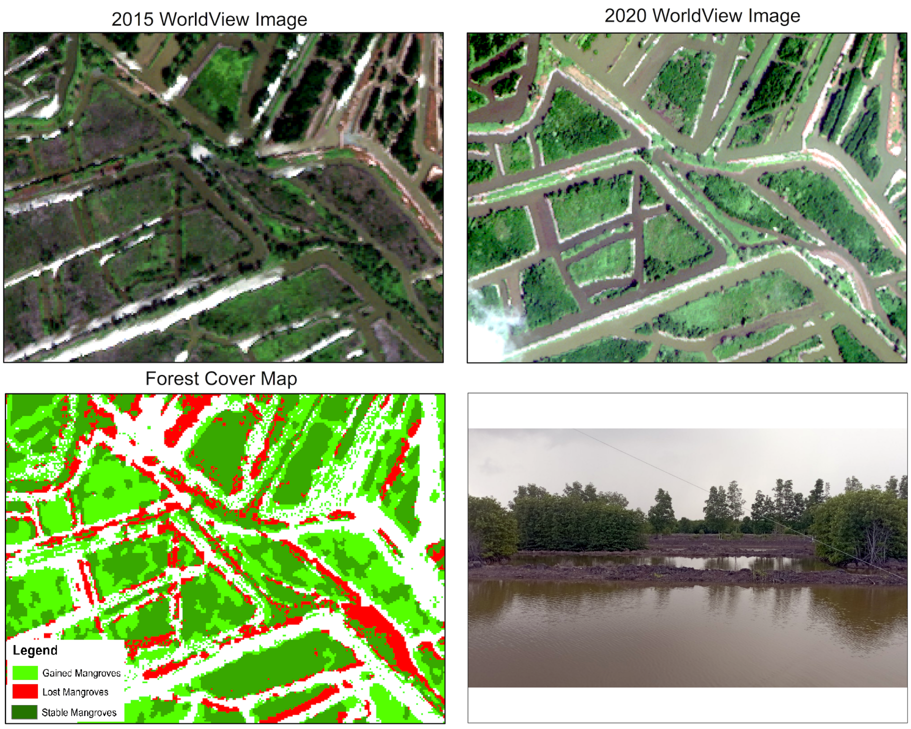


FIGURE 7
Mangrove loss in a integrated mangrove aquaculture area.

restoration is often the most likely to succeed (Lewis and Brown, 2014). It should be noted, however, that in this study we selected images taken near or at low tide to minimize the amount of mangrove area that was submerged by the tide, but that could not completely eliminate errors in mangrove classification in the intertidal mudflats where submerged mangrove forests locally occur even at low tide. More advanced techniques should be utilized to solve this issue in further studies.

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

PT and RM conceived the study, and wrote and edited the manuscript draft. PT, TH, HH and ML collected and analyzed satellite images, and performed data analysis. PT, TH, TV, PH, HH and ML collected field data. NH and BH assisted in gathering literature and writing the manuscripts draft. All authors contributed to the article and approved the submitted version.

Funding

This study was conducted with financial support from the US Forest Service International Programs (USFS/IP) under the project number IPC0.017.0911.001.

Acknowledgments

We want to thank the Polar Geospatial Center, University of Minnesota that kindly provided the WorldView-2 images. We also want to thank the local authorities for their supports in our field survey. Without their help and efforts, this work would not be possible.

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.

Publisher's note

All claims expressed in this article are solely those of the authors and do not necessarily represent those of their affiliated organizations, or those of the publisher, the editors and the reviewers. Any product that may be evaluated in this article, or claim that may be made by its manufacturer, is not guaranteed or endorsed by the publisher.

Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2022.1043943/full#supplementary-material

References

Alm, J., Masjkur, M., Arifatmi, B., Arquero, A., Artz, R., Ballhorn, U., et al. (2020). *Peatlands mapping and monitoring: Recommendations and technical overview* (Rome: Food and Agriculture Organization of the United Nations (FAO).

Alongi, D. M. (2012). Carbon sequestration in mangrove forests. *Carbon Manage.* 3, 313–322. doi: 10.4155/cmt.12.20

Binh, T. N. K. D., Vromant, N., Hung, N. T., Hens, L., and Boon, E. K. (2005). Land cover changes between 1968 and 2003 in cai nuoc, Ca mau peninsula, Vietnam. *Environ. Dev. Sustain.* 7, 519–536. doi: 10.1007/s10668-004-6001-z

Bullock, E. L., Fagherazzi, S., Nardin, W., Vo-Luong, P., Nguyen, P., and Woodcock, C. E. (2017). Temporal patterns in species zonation in a mangrove forest in the Mekong delta, Vietnam, using a time series of landsat imagery. *Continent. Shelf Res.* 147, 144–154. doi: 10.1016/j.csr.2017.07.007

Dan, T. T., Chen, C. F., Chiang, S. H., and Ogawa, S. (2016). Mapping and change analysis in mangrove forest. *ISPRS Ann. Photogramm. Remote Sens. Spatial Inf. Sci.* III-8, 109–116. doi: 10.5194/isprs-annals-III-8-109-2016

Das, S., Das, P., and Roy, B. R. (2020). "Cloud detection and cloud removal of satellite image–A case study". In: H. Sarma, B. Bhuyan, S. Borah and N. Dutta eds. *Trends in Communication, Cloud, and Big Data. Lecture Notes in Networks and Systems*, vol. 99. (Singapore: Springer). doi: 10.1007/978-981-15-1624-5_6

Dat, P. T., and Yoshino, K. (2011). "Monitoring mangrove forest using multi-temporal satellite data in the northern coast of Vietnam," In: *Proceeding of the 32th Asian Conference on Remote Sensing 2011*, 3-7 Oct. Taipei, Taiwan. 1, 169–174.

de Graaf, G. J., and Xuan, T. T. (1998). Extensive shrimp farming, mangrove clearance and marine fisheries in the southern provinces of Vietnam. *Mangroves Salt Marshes* 2, 159–166. doi: 10.1023/A:1009975210487

Donato, D. C., Kauffman, J. B., Mackenzie, R. A., Ainsworth, A., and Pfleeger, A. Z. (2012). Whole-island carbon stocks in the tropical pacific: Implications for mangrove conservation and upland restoration. *J. Environ. Manage.* 97, 89–96. doi: 10.1016/j.jenvman.2011.12.004

Fisher, J. R. B., Acosta, E. A., Dennedy-Frank, P. J., Kroeger, T., and Boucher, T. M. (2018). Impact of satellite imagery spatial resolution on land use classification accuracy and modeled water quality. *Remote Sens. Ecol. Conserv.* 4, 137–149. doi: 10.1002/rse2.61

Green, E. P., Clark, C. D., Mumby, P. J., Edwards, A. J., and Ellis, A. C. (1998). Remote sensing techniques for mangrove mapping. *International Journal of Remote Sensing* 19 (5), 935–56. doi: 10.1080/014311698215801

Hai, N. T., Dell, B., Phuong, V. T., and Harper, R. J. (2020). Towards a more robust approach for the restoration of mangroves in Vietnam. *Ann. For. Sci.* 77, 18. doi: 10.1007/s13595-020-0921-0

Halder, S., Samanta, K., Das, S., and Pathak, D. (2021). Monitoring the inter-decade spatial–temporal dynamics of the sundarban mangrove forest of India from 1990 to 2019. *Reg. Stud. Mar. Sci.* 44, 101718. doi: 10.1016/j.rsma.2021.101718

Hauser, L. T., Nguyen Vu, G., Nguyen, B. A., Dade, E., Nguyen, H. M., Nguyen, T. T. Q., et al. (2017). Uncovering the spatio-temporal dynamics of land cover change and fragmentation of mangroves in the Ca mau peninsula, Vietnam using multi-temporal SPOT satellite imagery, (2004–2013). *Appl. Geogr.* 86, 197–207. doi: 10.1016/j.apgeog.2017.06.019

Hawkins, S., To, P. X., Phuong, P. X., Thuy, P. T., Tu, N. D., Guong, C. V., et al. (2010). *Roots in the water: Legal frameworks for mangrove PES in Vietnam* (Washington, DC, USA: Forest Trends).

Hong, H. T. C., Avtar, R., and Fujii, M. (2019). Monitoring changes in land use and distribution of mangroves in the southeastern part of the Mekong river delta, Vietnam. *Trop. Ecol.* 60, 552–565. doi: 10.1007/s42965-020-00053-1

Hong, P. N., and San, H. T. (1993). *Mangroves of Vietnam* (Bangkok, Thailand: The IUCN Wetlands Programme).

Islam, M. M., Borgqvist, H., and Kumar, L. (2019). Monitoring mangrove forest landcover changes in the coastline of Bangladesh from 1976 to 2015. *Geocarto Int.* 34, 1458–1476. doi: 10.1080/10106049.2018.1489423

Kiprono, A. (2021). *An assessment of the effectiveness of mangrove restoration projects along the Kenyan coast* (Nairobi, Kenya: University of Nairobi).

Landis, J. R., and Koch, G. G. (1977). The measurement of observer agreement for categorical data. *Biometrics* 33, 159–174. doi: 10.2307/2529310

Lewis, R. R., and Brown, B. (2014). "Ecological mangrove rehabilitation – a field manual for practitioners," in *Mangrove action project* (Canadian International Development Agency and OXFAM).

Liu, S., Li, X., Chen, D., Duan, Y., Ji, H., Zhang, L., et al. (2020). Understanding land use/Land cover dynamics and impacts of human activities in the Mekong delta over the last 40 years. *Global Ecol. Conserv.* 22, e00991. doi: 10.1016/j.gecco.2020.e00991

Macintosh, D. J., and Ashton, E. C. (2003). *Draft code of conduct for the sustainable management of mangrove ecosystems* (Washington: World Bank, ISME, cenTER Aarhus).

MARD (2014). *Coastal forest protection and development plan to respond to climate change for the period 2015–2020.* (Hanoi, Vietnam: Vietnamese Ministry of Agriculture and Rural Development).

MARD (2021). *National program on protection and development of forests in coastal target to climate change and promote green growth for 2021–2030.* (Hanoi, Vietnam: Vietnamese Ministry of Agriculture and Rural Development)

Nam, V. N., Sasmito, S. D., Murdiyarso, D., Purbopuspito, J., and MacKenzie, R. A. (2016). Carbon stocks in artificially and naturally regenerated mangrove ecosystems in the Mekong delta. *Wetlands Ecol. Manage.* 24, 231–244. doi: 10.1007/s11273-015-9479-2

Nguyen, H. Q., and Brunner, J. (2011). *Land cover change assessment in the coastal areas of the Mekong delta 2004-2009* (Hanoi, Vietnam: IUCN).

Nguyen, H.-H., Nghia, N. H., Nguyen, H. T. T., Le, A. T., Ngoc Tran, L. T., Duong, L. V. K., et al. (2020a). Classification methods for mapping mangrove extents and drivers of change in thanh hoa province, Vietnam during 2005-2018. *For. Soc.* 4, 225–242. doi: 10.24259/fs.v4i1.9295

Nguyen, H. Q., Le, T. V. H., Quinn, C., Carrie, R., Pham, T. T. N., Stringer, L., et al. (2019). "Spatial planning influences mangrove forest development in Kim son district of Ninh Binh province". In: *FIG Working Week 2019: Geospatial information for a smarter life and environmental resilience.* (Hanoi, Vietnam).

Nguyen, D. N., and Nguyen, T. H. (2013). *Climate and climatic resources of Vietnam* (Hanoi, Vietnam: Publishing House of Science and Technology).

Nguyen, H.-H., Tran, L. T. N., Le, A. T., Nghia, N. H., Duong, L. V. K., Nguyen, H. T. T., et al. (2020b). Monitoring changes in coastal mangrove extents using multi-temporal satellite data in selected communes, Hai Phong city, Vietnam. *For. Soc.* 4, 256–270. doi: 10.24259/fs.v4i1.8486

Pham, T. T., Vien, N. N., Vo, Q. T., Tang, T. K. H., Nguyen, T. L., Tran, N. M. H., et al. (2022). *Opportunities and challenges for mangrove restoration in the Mekong delta: Status, policies and stakeholder* (Bogor, Indonesia: CIFOR).

Pham, T. D., and Yoshino, K. (2016). Impacts of mangrove management systems on mangrove changes in the northern coast of Vietnam. *Tropics* 24, 141–151. doi: 10.3759/tropics.24.141

Phan, M. H., and Stive, M. J. F. (2022). Managing mangroves and coastal land cover in the Mekong delta. *Ocean Coast. Manage.* 219, 106013. doi: 10.1016/j.ocecoaman.2021.106013

Phan, L. K., Jaap, S. M., van Thiel de Vries Marcel,, and Marcel, J.F.S. (2015). Coastal Mangrove Squeeze in the Mekong Delta. *Journal of Coastal Research* 31 (2), 233–243. doi: 10.2112/JCOASTRES-D-14-00049.1

SNV (2016). *MAM 1: Promoting ecosystem-based adaptation through reforestation and sustainable use of mangroves* (The Hague: Netherlands Development Organization).

Thakur, S., Maity, D., Mondal, I., Basumatary, G., Ghosh, P. B., Das, P., et al. (2021). Assessment of changes in land use, land cover, and land surface temperature in the mangrove forest of sundarbans, northeast coast of India. *Environ. Dev. Sustain.* 23, 1917–1943. doi: 10.1007/s10668-020-00656-7

Thomas, N., Bunting, P., Lucas, R., Hardy, A., Rosenqvist, A., and Fatoyinbo, T. (2018). Mapping mangrove extent and change: A globally applicable approach. *Remote Sensing* 10. 1466.

Thu, P. M., and Populus, J. (2007). Status and changes of mangrove forest in Mekong delta: Case study in tra vinh, Vietnam. *Estuarine Coast. Shelf Sci.* 71, 98–109. doi: 10.1016/j.ecss.2006.08.007

Tinh, P. H., MacKenzie, R. A., Hung, T. D., Hanh, N. T. H., Hanh, N. H., Manh, D. Q., et al. (2022). Distribution and drivers of Vietnam mangrove deforestation from 1995 to 2019. *Mitigation Adapt. Strateg. Global Change* 27, 29. doi: 10.1007/s11027-022-10005-w

Tong, P., Auda, Y., Populus, J., Aizpuru, M., Habshi, A., and Blasco, F. (2004). Assessment from space of mangroves evolution in the Mekong delta, in relation to extensive shrimp farming. *Int. J. Remote Sens.* 25, 4795–4812. doi: 10.1080/01431160412331270858

Toosi, N. B., Soffianian, A. R., Fakheran, S., Pourmanafi, S., Ginzler, C., and Waser, L. T. (2019). Comparing different classification algorithms for monitoring mangrove cover changes in southern Iran. *Global Ecol. Conserv.* 19, e00662. doi: 10.1016/j.gecco.2019.e00662

Tri, N. H. (1999). *Ecology of mangroves* (Hanoi, Vietnam: Agriculture Publishing House).

Truong, T. D., and Do, L. H. (2018). Mangrove forests and aquaculture in the Mekong river delta. *Land Use Policy* 73, 20–28. doi: 10.1016/j.landusepol.2018.01.029

Truong, S. H., Ye, Q., and Stive, M. J. F. (2017). Estuarine mangrove squeeze in the Mekong delta, Vietnam. *J. Coast. Res.* 33, 747–763. doi: 10.2112/JCOASTRES-D-16-00087.1

UNDP (2015). *Funding proposal: Improving the resilience of vulnerable coastal communities to climate change related impacts in Vietnam.* (Hanoi, Vietnam).

Veettil, B. K., Ward, R. D., Quang, N. X., Trang, N. T. T., and Giang, T. H. (2019). Mangroves of Vietnam: Historical development, current state of research and future threats. *Estuarine Coast. Shelf Sci.* 218, 212. doi: 10.1016/j.ecss.2018.12.021

Ward, R. D., Friess, D. A., Day, R. H., and Mackenzie, R. A. (2016). Impacts of climate change on mangrove ecosystems: A region by region overview. *Ecosyst. Health Sustain.* 2, e01211. doi: 10.1002/ehs2.1211

WB (2017). *Vietnam - forest sector modernization and coastal resilience enhancement project* (Washington: World Bank).

Xia, Q., Qin, C.-Z., Li, H., Huang, C., and Su, F.-Z. (2018). Mapping mangrove forests based on multi-tidal high-resolution satellite imagery. *Remote Sens.* 10, 1343. doi: 10.3390/rs10091343

frontiers | Frontiers in Marine Science

TYPE Policy and Practice Reviews
PUBLISHED 04 January 2023
DOI 10.3389/fmars.2022.1015262

Check for updates

OPEN ACCESS

EDITED BY
Richard MacKenzie,
Institute of Pacific Islands Forestry (USDA), United States

REVIEWED BY
Irwan Effendi,
Riau University, Indonesia
Harsuko Riniwati,
University of Brawijaya, Indonesia

*CORRESPONDENCE
Chen Mianrun
cmrandy@foxmail.com

SPECIALTY SECTION
This article was submitted to Marine Conservation and Sustainability, a section of the journal Frontiers in Marine Science

RECEIVED 09 August 2022
ACCEPTED 04 November 2022
PUBLISHED 04 January 2023

CITATION
Shusheng X, Jingqian X and Mianrun C (2023) Integrated management improves emerging coastal industries and ecological restoration with the participation of social capital. *Front. Mar. Sci.* 9:1015262. doi: 10.3389/fmars.2022.1015262

Integrated management improves emerging coastal industries and ecological restoration with the participation of social capital

Xu Shusheng[1], Xie Jingqian[2] and Chen Mianrun[1]*

[1]South China Sea Institute of Planning and Environmental Research, State Oceanic Administration, Southern Marine Science and Engineering Guangdong Laboratory (Zhuhai), Guangzhou, China,
[2]South China Sea Marine Survey and Technology Center, State Oceanic Administration, Guangzhou, China

In this study, we aimed to provide policy advice that supports continuous ecological restoration and coastal economic development. Our analysis indicated that insufficient funds and space are the main problems in mangrove restoration projects in China and Southeast Asian countries. The average cost of mangrove restoration projects in China has been RMB 999,000/ha, leaving a mangrove restoration funding gap of RMB 1,500,000,000/year. Another common problem of emerging industries is insufficient space, another is a lack of subsidies. Learning from the positive experiences of inland areas and Southeast Asian countries, we propose a plan for integrated management that improves emerging marine industries and ecological restoration with the participation of social capital. We also designed a time road map to achieve the plan based on a target area. A SWOT (Strengths, Weaknesses, Opportunities, and Threats) analysis showed that the plan is a win-win model plan, which may generally meet the needs of the local government, such as ecological restoration, pollution control, industrial upgrades, and income improvement. Finally, we suggest that governments should strengthen cross-department coordination, improve current sea area use policies, and strengthen associated publicity attempts.

KEYWORDS

mangrove ecological restoration, space resource utilization efficiency, emerging coastal industries, industrialized mariculture, ecotourism, integrated management, participation of social capital

Introduction

In recent decades, China's coastal areas have witnessed rapid economic development, accelerated population inflow, increased development intensity, and rapid accumulation of ecological dangers (Gao and Gou, 2014). Under the dual pressure of human activities and climate change, the coastal ecosystem is degrading. For example, the coastlines of some developed areas are seriously eroded; some typical marine ecosystems and habitats, such as mangroves, have been lost completely or are in an unhealthy state. In China, mariculture occupies a large amount of coastal space. However, mariculture farmers here usually own scattered mariculture ponds, hindering the organization of efficient production and upgrades of technology. Consequently, the scattered mariculture poses immense disadvantages, which influences the health of the coastal ecosystems. Additionally, because of antibiotic abuse and polluted aquaculture wastewater, aquaculture survival and growth rates are seriously restricted and the adjacent systems of coastal areas are degraded. In turn, the sustainable development of mariculture as well as the local economic development is negatively influenced. Therefore, pond mariculture must be upgraded to cleaner mariculture with better space utilization. They can also be improved by the introduction of fishery–solar complementary systems or use for tourism purposes. These emerging industries can help save space for ecological restoration by providing sustainable ecological services. Therefore, the combination of marine ecological restoration and upgradation of the mariculture industry will have immense practical significance.

Currently, the management, planning, and funding of these emerging industries and associated ecological restoration projects are set up separately and governed by different departments. For example, mariculture is governed by the agriculture and rural affairs department. Ecological restoration is governed by the natural resources department, ecotourism is managed by the culture and tourism department, and the management of fishery–solar complementary industries belongs to multiple departments, including the agriculture and rural affairs department and the national energy department. Different departments have different and possibly contradictory guidelines on managing the area. For example, even if a fishery–solar complementary company reaches an agreement with mariculture farmers, the latter may still have to obtain sea use permits. Such multiheaded management models cause low efficiency, ineffective utilization of space, and scattered application of funds. Furthermore, it discourages social investors from investing funds in these emerging industries. Moreover, in some ecologically degraded areas, such as mangrove forests, that have been subjected to mariculture activities for >40 years, the restoration plan not only requires a huge financial capital but also may pose a conflict with sustainable food supply and economic growth (Li et al., 2013; Yu et al., 2019; Xu et al., 2021b).

Overall, a new approach to economic development and ecological restoration is needed to achieve industrial upgrades in coastal areas with mariculture ponds. However, the development of emerging industries in these areas will encounter the problem of sea area use, considering that current sea area use policies make it difficult or even impossible to obtain relevant approvals (Xinhua News Agency, 2018). Thus, the need for space in the sea is vital for the ecological restoration and emerging industries in coastal mangrove forests.

To solve the problems discussed above, integrated management of ecological restoration efforts and upgrades in emerging industries should be practiced by the local government. To analyze a potential integrated management plan, we first illustrate the current status and problems of mangrove restoration projects and upgrades in local industries. We draw inspiration from the successful experience of Nanping city in China and some Southeast Asian countries. Finally, a SWOT (Strengths, Weaknesses, Opportunities, and Threats) analysis was performed on the proposed plan to present measures for attaining local sustainability.

Current situation of mangrove degradation

The total area covered by mangrove forests in China is ~27,100 hectares (BThe Third National Land Resource Survey, 2019). Mangroves in China are naturally distributed in southern provinces, including Guangdong, Guangxi, Fujian, Hainan, Taiwan, Hong Kong, and Macao. Mariculture has increased since the 1960s, and most mangroves in the middle and low intertidal zone were destroyed. By the 1990s, the damage worsened, and mangroves in the high intertidal zone were also destroyed, leading to serious degradation of coastal ecosystems. As Figure 1 shows, the mangrove area decreased by 62% from 1973 to 2000. Although the area covered has been restored to some extent since 2000, the area covered by mangroves by 2019 was still 44% lower than that in 1973 (Jia, 2014; Bureau of Nature Resources of Jiangmen City, 2019; Bureau of Statistics of Guangxi Zhuang Autonomous Region, 2019; Department of Nature Resources of Fujian Province, 2019 (unpublish); Department of Nature Resources and Planning, 2019 (unpublish); Xinhua News Agency (2021)).

Mariculture ponds (e.g., shrimp ponds) account for 97.6% of the area occupied by mangroves in China (Jia, 2014), severely reducing and damaging the mangrove ecosystems. On the one hand, high concentrations of untreated wastewater from these

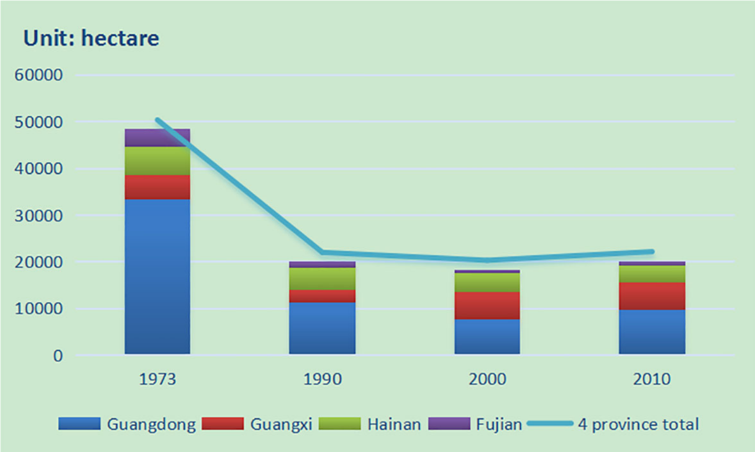


FIGURE 1
Change of mangrove area in China from 1973 to 2013 (Jia, 2014).

ponds are mostly directly discharged into remnant mangrove areas nearby, endangering the ecosystem further. On the other hand, fishery resources that rely on mangroves may also eventually be exhausted as mangroves die off, causing an unsustainable and unhealthy system of food production (Hu et al., 2012; Chatvijitkul et al., 2017; Hu et al., 2020).

Mariculture is similarly the main reason for mangrove loss in many Southeast Asian countries. In the Philippines, nearly half of the 279,000 hectares of mangroves that were lost from 1951 to 1988 were developed into mariculture ponds. The scale of mariculture ponds in Malaysia has gradually expanded since 1996, while the area covered by mangroves has decreased sharply or even disappeared. In Vietnam, the construction of shrimp ponds is one of the main reasons for the loss of mangroves; 42.5% and 60.1% of mangroves are estimated to be lost due to mariculture ponds in Nghe An and Thanh Hoa provinces, respectively (Hawkins et al., 2010; Pham et al., 2016; Nguyen et al., 2021).

History of mangrove restoration projects and associated problems

The degradation of mangroves causes many ecological problems, such as reduced ability to resist marine disasters, shrunken carbon sinks, and unsustainable fisheries (Bouillon et al., 2003; Gonneea et al., 2004; Alongi, 2012; Dung et al., 2016; Liang et al., 2018). China and some Southeast Asian countries have thus invested large sums of money in mangrove restoration projects. Although some projects accomplished their anticipated outcomes, most are still facing many problems.

Progress and development of mangrove ecological restoration projects

Ecological restoration of mangrove forests is highly valued in China. From 2010 to 2017, RMB ~13,700,000,000 were allocated by the central government of China from the special funds for marine ecological restoration. As a result, ~2,300 hectares of coastal wetlands have been successfully restored. These statistics do not include funds from local governments (Ministry of Nature Resources of People's Republic of China, 2018). Mangrove rehabilitation efforts account for the largest part of all marine ecological restoration projects in China and became the most focused supporting project in 2019. Particularly, the central government provided RMB 600,000,000 to five coastal provinces to implement mangrove rehabilitation in 2020 under the "Blue Bay" project. Meanwhile, "the special action plan for mangrove protection and restoration (2020–2025)" (hereinafter referred to as SAPMPR) was issued by the central government to continue its support for mangrove restoration. The SAPMPR aims to plant and restore 18,800 hectares of mangroves, including 9,050 hectares of new mangrove plantations and 9,750 hectares of existing mangrove restorations (Ministry of Natural Resources of People's Republic of China, 2020-4-9). These projects have achieved anticipated results as can be seen in Figure 1; from 2000 to 2019, the mangrove area has increased by 47%.

Similar to China, mangrove ecological restoration projects have been greatly supported by other countries in Southeast Asia. For example, Malaysia allocated 45,300,000 RM (RMB ~70,030,000) to implement a mangrove replanting program, along with other suitable tree specialties, along its national coastlines from 2005 to 2015. Vietnam planted 1,300 hectares of mangroves by 2020. Indonesia restored 37,539 hectares of mangroves by 2008 through its "Kebun Bibit Rakyat" project from 2010 to 2014, where mangroves were restored at a rate of 10,000 hectares per year (Hafsah, 2013).

Main problems faced by mangrove ecological restoration projects

The mangrove ecological restoration projects face two common problems in these countries: funds and spaces (Primavera and Esteban, 2008; Farley et al., 2009).

Insufficient funds and uneven distribution of funds

Although China has invested large funds in mangrove ecological restoration, they are still insufficient due to the huge restoration demand. Mangrove restoration projects are generally expensive, and the project costs vary in different regions. In some regions, the government must compensate pond owners for converting the ponds to mangrove forests. However, some owners may not be satisfied with the amount of compensation as mangrove ecological restoration projects lack practical economic profit. In addition, no profit leads to few participations in social capital. Table 1 lists the five mangrove restoration projects supported by China's central government in 2020. The average cost was RMB ~999,000/ha. As China's proposition for the "SAPMPR," according to our calculations, the capital needed is RMB ~18,600,000,000 for 5 years (RMB 3,100,000,000/year). However, in 2020, the capital used for mangrove ecological restoration was only RMB 600,000,000, and the funding gap was RMB 1,500,000,000 per year. The gap may be financed by local governments, leading to huge pressure on the local governments.

Overall, the funds for mangrove ecological restoration projects are unevenly distributed in space and time. For example, although the mangrove area of Guangdong is 1.12 times more than that of Guangxi, the funds given to Guangdong by the central government accounted for only 16.5% of the funds given to Guangxi in 2020. In addition, during the later stages of mangrove restoration projects, a lack of funds was very common. Most restoration projects received financial support within the first 1~3 years of promised funding only.

In Southeast Asia, there is a great funding gap for mangrove ecological restoration. Unlike in China, financial support from the central government is insufficient. Therefore, restoration projects in some Southeast Asian countries need international assistance. For example, the Global Environment Center (GEC) has funded many successful cases of mangrove ecological restoration in Malaysia. Vietnam received financial support from Japan in 2008, whereby a Japanese nongovernmental organization implemented a mangrove restoration project in the Thai Thui, Tien Lang, and Tinh GIO areas; a total of 1,100 hectares were planted. Vietnam also received 18,350,000 Euros of assistance from Germany and Australia in 2012 for the protection of mangrove wetlands and coastal areas, while the Vietnam government added another 2,600,000 Euros (BPham, T.D., and Yoshino, K., 2016).

Insufficient or unsuitable space

Although mangrove trees are flood-resistant plants, they cannot survive in severely flooded areas or areas without floods. It is thus difficult to find suitable places for mangrove forests (Yang, 2002). The SAPMPR undoubtedly has a huge demand for sea or land areas, which were located on the southeast coast of China and had high population densities with developed economies. In these areas, suitable places for mangrove restoration are very few, thus, artificially transformed coastal zones are the only choice, but they cannot offer high-

TABLE 1 Costs of comprehensive mangrove ecological restoration projects.

Year	Location	Investment (Unit: CNY)	Performance of mangrove restoration	Comprehensive average cost (Unit:CNY)
2020	Yangjiang city, Guangdong	76,811,300	Construct the cofferdam in the mangrove restoration area and the elevation reconstruction of the beach in the afforestation area.	384,100
2020	Fangchenggang city, Guangxi	253,530,900	Returning the dike to the sea and afforesting more than 100 hectares of mangroves	2,229,800
2020	Qinzhou city, Guangxi	211,721,700	More than 100 hectares of mangroves will be restored in Guangtan and other retirement areas. Restoration of natural mangrove plantations and secondary forest transformation.	1,085,800
2020	Wanning city	275,323,300	More than 300 hectares of aquaculture ponds were restored to mangrove forests	806,700
2020	Wenchang city	259,400,000	Restoration of mangrove forests	489,000
Total		1,076,787,200	1,379.89 hectares	999,000

Data Source: Local natural resources departments.

quality land for mangrove planting. Some mangrove restoration projects even replace other ecosystems, such as seagrass beds. For example, in a project implemented in Guangdong in 2012, seagrass beds were dug up to plant mangroves, which further degraded the coastal ecosystem.

The lack of space for mangrove restoration has also occurred in some countries in Southeast Asia. In the Philippines, although hundreds of millions of dollars have been invested in the restoration of mangroves in the past two decades, the long-term survival rate of mangroves is generally as low as 10%–20%. The main reason is inappropriate planting space, while the ideal space has been transformed into fishponds. Mangroves should be replanted where fishponds have replaced them, not where they never existed, such as seagrass beds and tidal flats (Primavera and Esteban, 2008; Farley et al., 2009).

Emerging industries in coastal zones

Emerging industries in coastal zones have recently developed, which include industrialized mariculture, fishery–solar complement, and marine ecotourism. These new industries may replace traditional industries and reduce sea area use.

Industrialized mariculture

Industrialized mariculture includes running water and circulating water mode. Its major characteristics are three-dimensional (3D) breeding, water quality, and disease control. It can increase output per unit area, stabilize the quality of products, and reduce the sea or land use area. Therefore, it features saved water, all-weather suitability, safe production methods, reduced pollution, and the option to choose different mariculture species (BTiller, 2015).

In China, industrialized mariculture has developed for more than 30 years. To speed up the development of industrialized mariculture, China has issued a policy to encourage the development of industrialized mariculture (Ministry of Agriculture and Rural Affairs of People's Republic of China, 2021).

The breeding of turbot (*Paralichthys olivaceus*), salmon (*Salmonidea*), and other species is common in China. The volume of China's industrialized mariculture was 39,410,000 m^3 in 2020 with a 12.1% growth year-on-year; the total output of mariculture products was 325,000 tons, and the output per unit volume was 8.3 kg/m^3. However, circulating water technology has been adopted by less than 15% of industrialized mariculture industries due to its high cost, complex process, high energy consumption, and requirements for technical skills and equipment. The biggest disadvantage is high cost, especially during the early stages of setup (Antonio et al., 2000; Blancheton, 2000; Suantika et al., 2001; Saidul et al., 2014; Sha and Zhu, 2021). Nowadays, industrialized mariculture factories in southern China are less than those in northern China because it is more profitable to implement them in the adjustable temperature of colder areas.

Fishery–solar complementary industries

The "fishery–solar complementary" industries aim to build solar power generation machines on the water surface of mariculture ponds. They generate electricity simultaneously with mariculture, which saves coastal space resources. Additionally, it provides clean energy (Wang, 2017; Guo et al., 2017; Lei and He, 2021), improves the environment, and is of great significance for carbon neutralization. The electricity produced in the mangrove forests of Southeast China can be consumed locally by factories, which reduces wasted electricity as compared to the transportation of electricity from western China. In China, policies promoting the development of the fishery–solar complementary industries have been in place, attracting the attention of social capital investors, such as energy companies.

For this emerging industry, the difficulty is not a lack of social capital but the difficulty of obtaining permissions. For example, investors should sign agreements with pond owners, which requires negotiation. Additionally, investors must apply for governmental permission to use the sea area, which is open for mariculture ponds only in some areas. This process may include procedures to obtain permits from other government departments, such as the Forestry Department and the Environmental Protection Department.

Marine ecotourism

Marine ecotourism is a promising industry based on good marine ecology and well-protected marine culture. At present, China's marine tourism is booming, as the tourism industry is extending from land to sea (Liu, 2012; Chen et al., 2020; Zheng and Zhu, 2020; Su et al., 2021). However, marine ecotourism is threatened or restricted; for example, the natural landscape has been destroyed by a large area of mariculture ponds and consequential pollution (BQiu et al., 2018; BWang, 2020). Tourism facilities are also damaged by natural disasters; marine ornamental animals disappear as biodiversity declines (Xu, 2012; Li and Yu, 2020; Wang, 2020).

Means of addressing the coastal ecological degradation and emerging industries

Coastal ecological restoration and emerging industries have interacted with each other for mutual betterment in different parts of the world, and thus, we must find inspiration from other countries and places.

The problems between ecological restoration and emerging industries

As shown in Figure 2, local governments face dual difficulties from the perspectives of economic development and ecological restoration. Due to the lack of social capital, the tax generated from traditional mariculture cannot provide sufficient funds for ecological restoration. Ecological restoration thus requires input from other taxes, such as transfer payment funds from the central government. However, this model is unsustainable. To continue ecological restoration consistently in the future and develop a sustainable economy for it, mariculture ponds must be upgraded to industrialized mariculture, which will offer numerous advantages, including compact structure, low pollution, and more efficiency. At the same time, the restored ecosystem can be used to develop ecotourism, such as leisure fishing. However, ecological restoration consumes a lot of space, resources, and taxes, making it difficult to attract social capital; meanwhile, emerging industries need these same resources (e.g., space) or start-up funds. If these emerging industries cannot be developed, the local ecological restoration will lose sustainable funds. Therefore, the core of the problem is that the transfer payment funds have not improved the efficiency of local space utilization, leaving the local economic output per unit area unimproved.

Inspiration from other countries and places

Ecological banks

An ecological bank is a platform that establishes cooperation between scattered farmers and social capital (Xu et al., 2021a; Department of Nature Resources of Fujian Province, 2020-5-8, 2020-5-8). In Fujian province, China, the ecological bank converts scattered resources into capital and exports ecological products. Furthermore, it offers a common platform that emphasizes market operation so that local governments, financial institutions, professional operators, and farmers can divide their work and cooperate (Yang and Guo, 2017; Huang et al., 2020; Gao et al., 2020; Zhang et al., 2020).

Experiences from other comprehensive projects in China

In China, the Zhanjiang city government found a balance between the mutual benefit of mangrove protection and enclosure

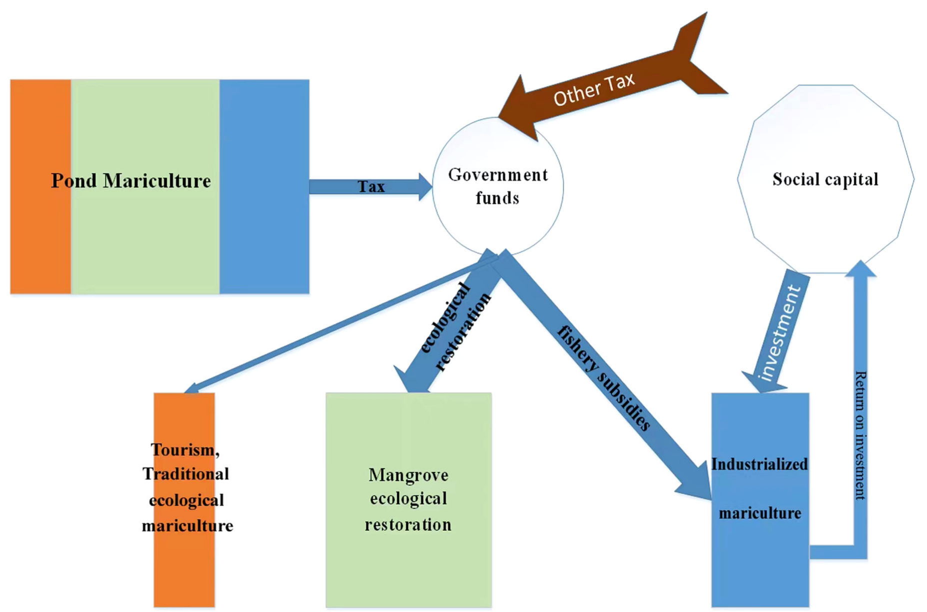


FIGURE 2
Current investment patterns of government funds and social capital.

mariculture practices. In 2021, they launched a pilot project that applied a comprehensive coupling-coexistence model to mangrove ecological restoration and mariculture ponds that not only protected and restored mangroves but also retained their economic benefits (Yang and Guo, 2017; Xinhuanet 2022).

Experiences from Southeast Asian countries

In Vietnam, the local government established shrimp mariculture and mangrove mixed forestry enterprises (SFMFEs) to reduce mangrove losses caused by shrimp ponds. The SFMFEs require workers to breed shrimp and plant mangroves (e.g., *Rhizophora apiculata*) at the same time (Hawkins et al., 2010).

In the Philippines, Professor Danao proposed the concept of "ecological symbiosis," which ensures the protection of mangroves within the overall design of mariculture (Danao, 2019). Ecological symbiosis offers water of better quality through biofiltration, and the existing fishponds are changed to eco-aquaculture zones. Production efficiency is improved by better management and use of technology while developing ecotourism and increasing the income of the community (Figure 3).

A proposed integrated management plan

Based on the analysis and inspiration, we propose a plan for "integrated management of emerging marine industries and ecological restoration with the participation of social capital" to improve both factors (Figure 4). The plan is as follows. Firstly, the local government will establish an "ecological bank," whose equity will come from capitalized ecological resources of scattered pond owners, financial funds of different government departments (including ecological restoration funds, environment protection funds, fishery subsidies, and emerging industry subsidies), and social capital. The task of the ecological bank is to reduce government expenditure, attract more social investment, and obtain policy and technical support. Secondly, different government departments will formulate a common development plan, whereby funds from them are concentrated to support emerging industries. Some profits from the emerging industries will be used to support new ecological restoration projects, while some will be returned to pond owners and social investors to complete the capital circulation. The restored ecological environment will eventually benefit both the emerging industries and pond owners. Specifically, coastal area "A" in Figure 5 was covered by mangroves, which were cut down to create mariculture ponds. Although these ponds contribute to taxes, they bring pollution and a decline in the ability to resist natural disasters, making the local economy unsustainable.

To implement the proposed plan, the capital value of every pond owner's ecological resource and production equipment will be evaluated; government funds from ecological restoration, fishery, and tourism development departments will be injected into this bank. After that, professional mangrove ecological restoration projects and industrialized mariculture projects can

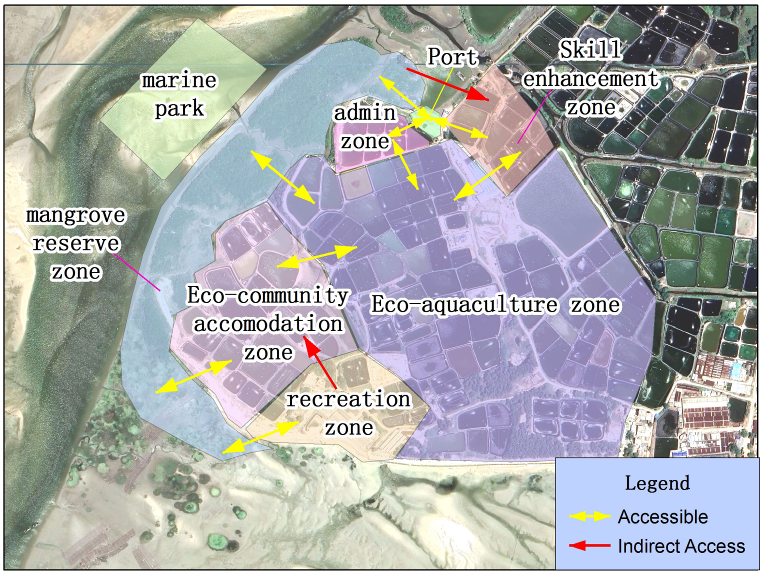


FIGURE 3
Overall design zoning map of Brakan mangrove protection and mariculture.

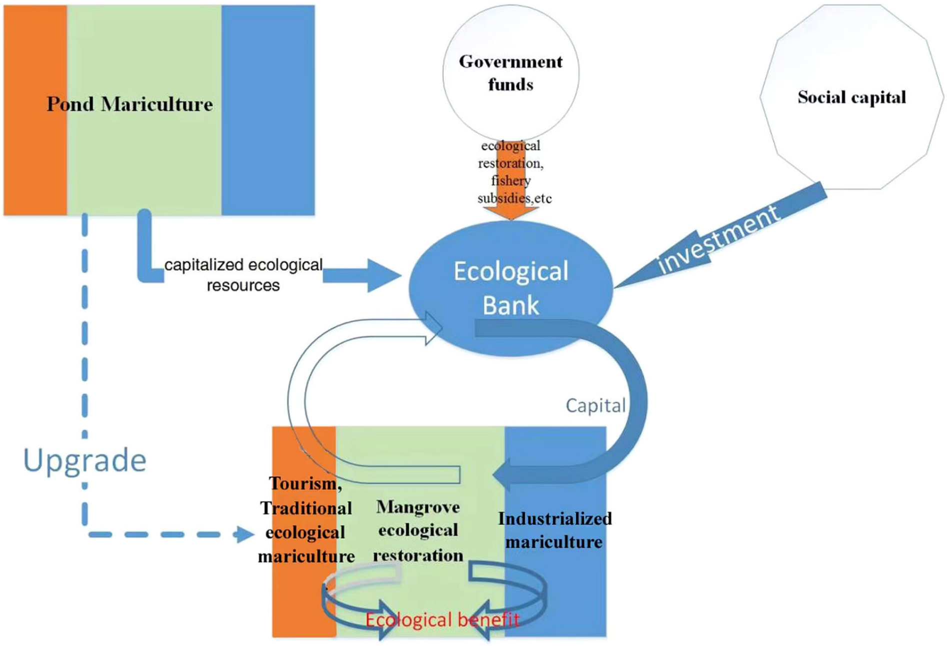


FIGURE 4
Model diagram after introducing the ecological bank.

be implemented while attracting social investment. The income of the pond owners will be stable, as they can work in the emerging industry being set up. For those pond owners who are unwilling to work in factories, work can be offered in ecological mariculture ponds that are less polluted, have higher biodiversity, and are a promising ecotourism resource (Feng et al., 2017). For example, old shrimp ponds can be transformed into typical "Kiwai ponds" in which adult shrimps are allowed to reproduce in summer, while the ponds are dry in autumn. The floodgates of Kiwai ponds will be opened during high tide to invite juvenile shrimps from the ocean naturally. A bird island can be set up in the middle of the Kiwai ponds, allowing the migratory birds to eat and breed in the area (Jacob et al., 2018). The Kiwai ponds can produce higher-quality shrimp. Industrialized mariculture can thus save a lot of coastal space while allowing fishery–solar complement to be implemented simultaneously, whereby solar panels are installed on and around the factories. At the same time, part of the space saved and profits earned by emerging industries can be used for mangrove ecological restoration. When the mangrove forests take root, they will bring benefits to the locals, such as habitat restoration of fishery seedlings, marine disaster reduction, and carbon sequestration. All of these profits can in turn save local government financial expenditure and increase the income of mariculture farmers and emerging industry investors.

SWOT (Strength, Weak, Opportunity, Threat) analysis of the proposed management approach

In 1982, Dr. Heinz Weihrich proposed the SWOT matrix analysis method. We performed the SWOT analysis to study the feasibility of the proposed plan for integrated management. The analysis covers the situation, internal conditions, and external competition. It lists and arranges internal strengths (S), weaknesses (W), external opportunities (O), and threats (T) of a proposed object in the form of a matrix—QCDMS (quality, cost, delivery, safety, and morale)—then it is listed for an internal analysis, and then PEST (political, economic, social, and technological) is listed for an external analysis. All of the factors are listed in internal strategic factor analysis (IFAS) and external strategic factor analysis (EFAS) tables. The factors with direct, important, massive, urgent, and long-term influence on the implementation of the plan are listed first, while those with indirect, secondary, minor, unimportant, and short-term influence are listed later. The possible score ranges from 5 to 1, and the sum of all IFAS and EFAS weights is 1 each. All factors are sorted according to the expert scoring method. By combining internal factors with the external environment for matrix analysis, four main strategies (SO, WO, ST, and WT) were obtained to cope with environmental changes (Sayyed et al.,

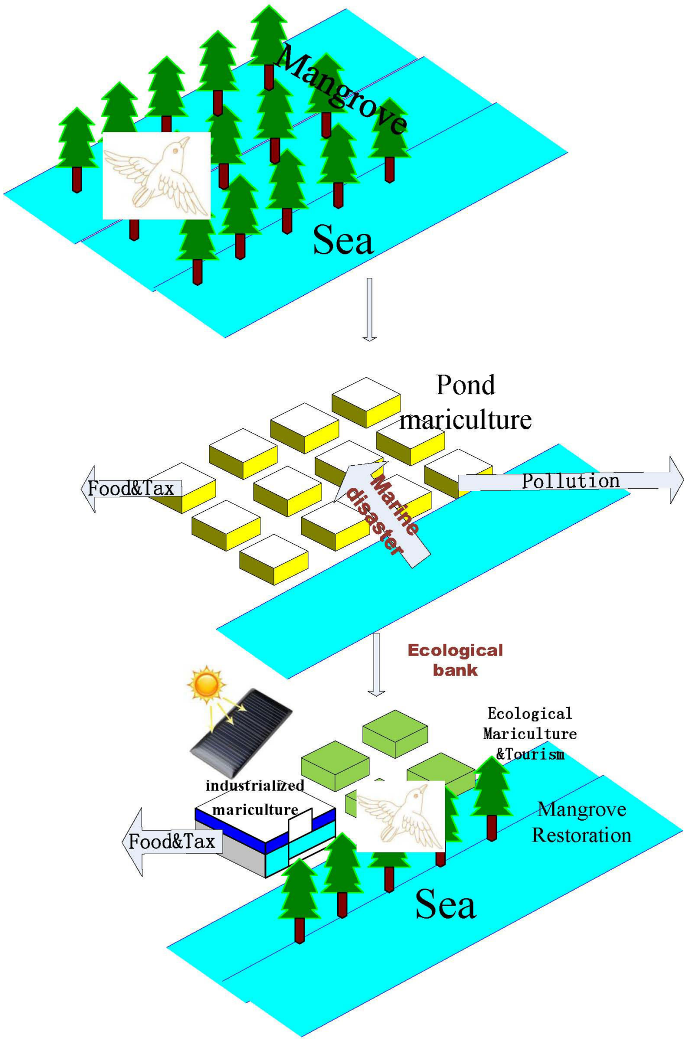


FIGURE 5
Sketch map of the comprehensive project of ecological restoration and green mariculture.

2013; Rong and Rui, 2015; Rezazadeh et al., 2017; Roslan et al., 2018; Sartor et al., 2019).

Internal strategic factor analysis

Strength analysis (S)

1) The ecological and economic quality in coastal areas was improved. Firstly, information related to funds for mangrove restoration is needed by many stakeholders, especially the government. The proposed plan can collect different information to meet the needs of more stakeholders. Secondly, the use of antibiotics can be greatly reduced as the circulating water method is used. Meanwhile, large-scale mariculture factories can integrate with new technology, such as Internet of Things (IoT), making products traceable and their quality more transparent. At last, industrialized mariculture factories can reduce the discharge of polluted water. Overall, traditional Kiwai ponds and restored mangrove wetlands can improve water quality and biological diversity.

2) The cost of government and enterprises is reduced. Part of the funds for early ecological restoration will be used to subsidize emerging industries and reduce their overall

costs. Emerging industries can then attract more social capital, which can partly be used for ecological restoration, reducing fiscal expenditure by the government.

3) The efficiency of space utilization is improved. The plan can transform the single mariculture pond space into various spaces, such as spaces for 3D industrialized aquaculture, ecological restoration, ecotourism, and new mangrove plantations, which would improve the efficiency of space utilization. These setups will be supported by government policy.
4) The efficiency of talents and equipment is improved. Talents and human resources will be gathered from all disciplines to implement the plan and integrate a variety of equipment, such as construction machinery, to improve the efficiency of the personnel.
5) More sustainable ecological services are provided. With the participation of social capital, follow-up funds for ecological restoration are guaranteed. Three-dimensional mariculture factories can provide a stable output of products and reduce the pressure on local coastal development, while the mangroves can provide more sustainable ecological services.

Weakness analysis (W)

1) The reliability of the plan must be tested. Compared to single projects developed independently, the proposed plan has no predecessor at present and may face unexpected policy problems. An important risk to avoid is factory mariculture companies defrauding government subsidies in the name of ecological restoration.
2) The cost is higher in the early stage. The initial investment includes demolition compensation for pond owners, construction of mariculture factories, and investment in ecological restoration, which may require more start-up funds. This means that a greater proportion of funds must be invested by the government during the early stage.
3) Technical defects. Circulating water machines need a continuous energy supply; in case of a power failure, a large number of fish and shrimp may die. The efficiency of solar power generation is not very high, and it faces the challenges of moisture, salt fog, and natural disasters near the sea.
4) The operational process of equipment is more complex. The operation of industrialized mariculture and other equipment is complex; therefore, workers must be professionally trained. Conventional mariculture farmers may not be suitable for the jobs pertaining to the proposed project.
5) Standardization of the emerging industries and ecological restoration need to be strengthened. The standardization of the operation of industrial mariculture facilities and equipment, fishery–solar complementary, ecotourism, fund subsidy, and the ratio of ecological restoration funds/profits of emerging industries need to be established and strengthened.

External strategic factor analysis

Opportunity analysis (O)

1) Governmental policies are supportive of emerging industries and ecological restoration. Local governments face pressure from mangrove planting tasks, such as insufficient space for suitable mangrove plantations and compensation funds for pond withdrawal. Historically, under the pressure of ecological restoration needs, some local governments have had to choose areas without natural mangroves, resulting in new ecological problems. To solve these problems, the national and local governments must be supportive of a replacement plan for ecological restoration.
2) Local economic growth needs the development of emerging industries, while ecological restoration needs sustained financial support as well. The local government faces the pressure of economic development. However, traditional mariculture has been sluggish, and the development of emerging industries is needed. Mangrove ecological restoration implemented until now has suffered from insufficient follow-up funds. Single projects especially generally face financial problems. Due to the lack of profit, mangrove ecological restoration projects implemented lack management as well.
3) Society needs greater quality ecological services and mariculture products. Many mangroves are surrounded by shrimp ponds, even in mangrove reserves. Before such reserves were established, shrimp ponds already existed, and the quality of shrimp was not good due to excessive antibiotic application. In contrast, the social demand for ecological services and mariculture products is now growing.
4) Factory mariculture and solar energy are recognized as major development directions for modern industry. Industrial mariculture, solar power generation, and

other coastal emerging industries are recognized as the main development direction for modern industries because of their many technical advantages.

Threat analysis (T)

1) The plan may risk policies on sea area use. Some mariculture ponds suffer from historical issues of reclamation, where the pond owners did not obtain registration certificates for sea area use. Once the construction of mariculture factories and the development of ecotourism start, the right to use these areas must be confirmed. However, under the current sea area use policy, it is difficult to obtain administrative permissions due to strict controls.
2) It is difficult to coordinate across different government departments. At present, projects on mangrove ecological restoration, industrialized mariculture, and ecotourism are managed by different government departments, such as the natural resources, agriculture, tourism, and energy departments. Governmental financial funds are applied for and distributed by these departments, while some originate from the central and local governments, making them difficult to be coordinated.
3) Stakeholders need to be coordinated. Mariculture farmers and community residents need to be involved in coordinated efforts. The proposed plan may result in changing the lifestyle of local mariculture farmers, which may not be accepted by all.
4) The spread of technology in emerging industries is not wide enough. Emerging industries need a large number of workers with suitable professional skills; however, most local mariculture farmers do not master these skills. Meanwhile, technical problems in emerging industries must be solved, but there is a lack of training in this regard.

Finally, we sorted the analyzed contents according to priority and degree of influence. Three experts scored and weighted every factor, and the final score was calculated (Figure 6). From the IFAS, we can see that the plan's score was 3.85, which is higher than the average. Therefore, the plan can offer advantages and possibly avoid its disadvantages. From the EFAS, a higher-than-average score of 4.27 was calculated, showing that the plan can readily avoid external threats and seize opportunities. By combining the internal factors with the external environment for matrix analysis, four main strategies were obtained to cope with environmental changes (Figure 7). The scores of S, W, O, T are 2.41, 1.44, 2.74, 1.53, respectively, so quadrant 1 scored highest, and the SO strategy will be the best choice.

Conclusions and recommendations

We performed an analysis of the current situation and existing problems pertaining to ecological restoration and emerging

		Content	Score	Weight	No		Content	Score	Weight	No	Total
IFAS	S	The ecological and economic quality in coastal areas is improved	4.11	0.14	0.58	W	The reliability is need to be tested	4.00	0.10	0.40	3.85
		The cost of government and enterprises is reduced	4.89	0.12	0.59		The cost is higher in the early stage	3.78	0.11	0.42	
		The efficiency of space utilization is improved	4.89	0.10	0.49		Technical defects	3.44	0.08	0.28	
		The efficiency of talents and equipments is improved	3.44	0.09	0.31		Operation process of equipments are more complex	2.78	0.08	0.22	
		More sustainable ecological services is provided	4.44	0.10	0.44		Standardization need to be strengthened	2.56	0.05	0.13	
EFAS	O	The government's policies are supportive	4.89	0.18	0.88	T	Sea area use policy risk	4.11	0.15	0.62	4.27
		Local economic growth and ecological restoration needs capital	4.89	0.20	0.98		It is difficult to coordinate across different government departments	4.11	0.10	0.41	
		Society needs higher quality ecological services and mariculture products	4.33	0.12	0.52		Stakeholders need to be coordinated	3.33	0.08	0.27	
		Main development direction of modern industry	4.00	0.09	0.36		The spread of emerging industry technology is not wide enough	3.00	0.08	0.24	

FIGURE 6
A SWOT (Strengths, Weaknesses, Opportunities, and Threats) analysis of the integrated management of ecological restoration and emerging industries.

Internal / Strategy / External	S	W
	The cost of government and enterprises is reduced. The efficiency of space utilization is improved. More sustainable ecological services are provided. The ecological and economic quality in coastal areas is improved. The efficiency of talents and equipment is improved.	The reliability must be tested. The cost is higher during the early stages. Technical defects must be considered. Operational processes of equipment are more complex. Standardization must be strengthened.
O The governmental policies are supportive. Local economic growth and ecological restoration need capital. Society needs high-quality ecological services and mariculture products. The major development direction of modern industry is the fisheries-solar complement.	**SO** Strengthening the cooperation of relevant departments in ecological civilization construction Promoting the policies of sea area and land use for ecological restoration to be readjusted. Strengthen publicity, attract more social investment, carry out "green finance", and establish a sustainable fund for ecological restoration.	**WO** Increase the government's financial support, especially during the early stages of the implementation of the proposed plan. Speed up the elimination of old industries.
T There is a risk to policy on sea area use. It is difficult to coordinate across different government departments. Stakeholders need to be coordinated. The spread of emerging industries and their technology is not wide enough.	**ST** Strengthening the government's position and clarifying firm regulations. Community should be educated to realize the importance of mangroves for all levels of society and to enhance their skills to join emerging industries.	**WT** Strengthen publicity by inviting all stakeholders and the community to jointly care about the restoration of coastal development. Develop emerging industries that have little impact on existing industries, such as fishery-solar complements and eco-tourism.

FIGURE 7
Matrix analysis of the internal elements with the external environment.

industries in coastal zones; lessons were gathered from the experiences of regions in China and Southeast Asian countries. We propose a plan to integrate mangrove restoration projects and the activities of emerging industries. A SWOT analysis was performed to understand the strengths, weaknesses, opportunities, and threats of the plan to obtain the SO, ST, WO, and WT strategies. Our results showed that the proposed plan can meet the needs of local governments, such as pollution control, industrial upgrades, and improvement in residents' income. The plan will also ensure follow-up funds for mangrove ecological restoration while providing land for emerging industries. After the successful implementation of the plan, the quality of mariculture products is expected to be improved, and additional employment opportunities are guaranteed for the locals. The plan is highly consistent with national government policies, making it a win-win model. For the optimal implementation of the plan, the advice extracted from the SWOT analysis should be considered. With these strategies, it is hoped that problems, such as insufficient funds and space and the need for emerging industrial development, will begin to be addressed.

Author contributions

XS and CM write the paper together, the work was supported by CM's grants. XJ have drawled figures, and analyze data. All authors contributed to the article and approved the submitted version.

Funding

This work was supported by the Innovation Group Project of Southern Marine Science and Engineering Guangdong Laboratory (Zhuhai) (No.311021004) and Southern Marine Science and Engineering Guangdong Laboratory (Zhuhai) (SML2021SP308), which are led by CM.

Acknowledgments

We thank to the Innovation Group Project of Southern Marine Science and Engineering Guangdong Laboratory (Zhuhai) (No.311021004)and Southern Marine Science and Engineering Guangdong Laboratory (Zhuhai) (SML2021SP308) for fund supporting and research findings sharing.

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.

Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2022.1015262/full#supplementary-material

References

Alongi, D. M. (2012). Carbon sequestration in mangrove forests. *Carbon Manage.* 3 (3), 313–322. doi: 10.4155/cmt.12.20

Antonio, T., Carlos, M., and Manuel, P. (2000). Environmental impacts of intensive aquaculture in marine waters. *Water Res.* 34 (1), 334–342. doi: 10.1016/S0043-1354(99)00102-5

Blancheton, J. P. (2000). Developments in recirculation systems for Mediterranean fish species. *Aquacult Eng.* 22 (1-2), 17–31. doi: 10.1016/S0144-8609(00)00030-3

Bouillon, S., Dahdouh-Guebas, F., and Rao, A. (2003). Sources of organic carbon in mangrove sediments: variability and possible ecological implications. *Hydrobiologia* 495 (1), 33–39. doi: 10.1023/A:1025411506526

Bureau of Nature Resources of Jiangmen City *Main data bulletin of the third national land survey of guangdong province*. Available at: http://www.jiangmen.gov.cn/jmzrj/gkmlpt/content/2/2507/post_2507058.html#187.

Bureau of Statistics of Guangxi Zhuang Autonomous Region *Main data bulletin of the third national land survey of guangxi*. Available at: http://tjj.gxzf.gov.cn/zxfb/t10390768.shtml.

Chatvijitkul, S., Boyd, C. E., and Davis, D. A. (2017). Pollution potential indicators for feed-based fish and shrimp culture. *Aquaculture* 477, 43–49. doi: 10.1016/j.aquaculture.2017.04.034

Chen, N., Liu, J., Ba, Z. X., Zhong, J. F., and Liu, X. L. (2020). The construction and research of marine tourism management system based on the perspective of industrial integration. *J. Coast. Res.* 112, 132–135. doi: 10.2112/JCR-SI112-038.1

Danao, M. P. (2019). "Ecological symbiosis-integrated design method of brakan mangrove protection and mariculture," in *Proceedings of 2019 Hengshui Lake Ecological Civilization International Exchange Conference*. 90–95.

Department of Nature Resources of Fujian Province *Main data bulletin of the third national land survey of fujian province*. Available at: http://zrzyt.fujian.gov.cn/zwgk/zfxxgkzl/zfxxgkml/tdgl_19753/202112/t20211231_5805488.htm.

Department of Nature Resources of Fujian Province (2020) *Good news-nanping "forest ecological bank" was selected as the first batch in China typical cases of value realization of ecological products*. Available at: http://sthjt.fujian.gov.cn/zwgk/ywxx/stbh/202005/t20200508_5261709.htm.

Dung, L. V., Tue, N. T., Hieu, P. V., Quy, T. D., and Omori, K. (2016). Tracing organic carbon dynamics in mangrove ecosystems using stable isotope analysis. *Proc. Int. Symposium Hanoi Geoengineering* 95, 53–58. doi: 10.1016/j.aquabot.2007.12.005

Farley, J., Batker, D., de la Torre, I., and Hudspeth, T. (2009). Conserving mangrove ecosystems in the Philippines: transcending disciplinary and institutional borders. *Environ. Manage* 45 (1), 39–51.

Feng, J. X., Zhu, X. S., Ning, C. X., Lin, Q. X., and Lin, G. H. (2017). Evaluation of mangrove-aquaculture integrated system in wetland eco-restoration in shenzhen. *China Environ. science.* 37 (7), 2662–2673.

Gao, Q., and Gou, L. (2014). Research on the marine ecological restoration in the industry substitution theory. *Sci. Manage.*, 34 (06), 61–65.

Gao, X. L., Lin, Y. Q., Xu, W. H., and OUYANG, Z. Y. (2020). Research progress on the value realization of ecological products. *Acta Ecologica Sin.* 40 (1). doi: 10.5846/stxb201807211563.

Gonneea, M. E., Paytan, A., and Herrera-Silveira, J. A. (2004). Tracing organic matter sources and carbon burial in mangrove sediments over the past 160 years. *Estuar. Coast. Shelf Sci.* 61 (2), 211–227. doi: 10.1016/j.ecss.2004.04.015

Guo, L. C., Hu, X. D., Su, Y. Y., Wu, P. P., and Weng, S. G. (2017). Design and flood control assessment of 5MWp fishing and photovoltaic power project in xinghua city. *IOP Conf. Series: Earth Environ. Sci.* 100 (1), 1–5. doi: 10.1088/1755-1315/100/1/012034

Hafsah, E. (2013). Implementasi program kebun bibit rakyat (KBR). *Jurnal Demokrasi Dan Otonomi Daerah* 3, 33–36.

Hawkins, S., To, P. X., Phuong, P. X., Thuy, P. T., and McNally, R. (2010). Roots in the water: Legal frameworks for mangrove PES in vietnam. katoomba group's legal initiative country study series. *For. Trends* 20, 19–43.

Huang, Y., Wen, T. J., Fan, S. S., and Luo, C. L. (2020). Economics of scale, multi-incentives and realization of ecological product value: Summary of experience of forest eco-bank in nanping, fujian. *Issues Forestry Economics* 40 (5), 499–509. doi: 10.16832/j.cnki.1005-9709.20190361

Hu, Z., Lee, J. W., Chandran, K., Kim, S., and Khanall, S. K. (2012). Nitrous oxide (N2O) emission from aquaculture: A review. *Environ. Sci. Technol.* 46 (12), 6470. doi: 10.1021/es300110x

Hu, B. B., Xu, X. F., Zhang, J. F., Wang, T. L., Meng, W. Q., and Wang, D. Q. (2020). Diurnal variations of greenhouse gases emissions from reclamation mariculture ponds. *Estuarine Coast. Shelf Sci.* 237, 37–40. doi: 10.1016/j.ecss.2020.106677

Jacob, C., Buffard, A., Pioch, S., and Thorin, S. (2018). Marine ecosystem restoration and biodiversity offset. ecological engineering. *J. Ecotechnol*, 120, 585–594. doi: 10.1016/j.ecoleng.2017.09.007

Jia, M. M. (2014). *Remote sensing analysis of mangrove dynamic changes in China from 1973 to 2013* (Hefei City, Anhui province, China:University of Chinese Academy of Sciences).

Lei, Z., and He, Y. C. (2021). Kong B solar radiation influence on power generation benefit of large-scale fishery solar complementary photovoltaic power station. *J. Nanjing Univ. Inf. Sci. & Technol(Natural Sci. Edition)* 13 (3), 377–382. doi: 10.13878/j.cnki.jnuist.2021.03.016

Liang, Y. X., Cheng, X. W., Zhu, H., Shutes, B., Yan, B. X., Zhou, Q. W., et al. (2018). Historical evolution of mariculture in China during past 40 years and its impacts on eco-environment. *Chin. Geographical Sci.* 28 (3), 363–373. doi: 10.1007/s11769-018-0940-z

Li, N., Chen, P. M., Qiao, P. P., and Qin, C. X. (2013). Research advances on marine ecological effect and repair techniques of coastal mangrove wetland. *Guangdong Agric. Sci.* 40 (20), 157–160, 167. doi: 10.3969/j.issn.1004-874X.2013.20.047

Liu, H. (2012). Problems and countermeasures faced by china's marine ecotourism industry. *Acad. Theory* 13), 93–94.

Li, D. M., and Yu, X. Y. (2020). Analysis and optimization of influencing factors of zhuhai marine tourism economic development. *J. Coast. Res.*, 106 (SI), 101–104. doi: 10.2112/SI106-026.1

Ministry of Agriculture and Rural Affairs of People's Republic of China (2021) *Opinions of the ministry of agriculture and rural affairs on accelerating the development of aquaculture mechanization*. Available at: http://www.moa.gov.cn/nybgb/2020/202012/202102/t20210201_6360857.htm.

Ministry of Nature Resources of People's Republic of China (2018) *China Continues to strengthen marine ecological restoration*. Available at: http://www.gov.cn/xinwen/2018-07/27/content_5309754.htm.

Ministry of Nature Resources of People's Republic of China (2020) *The ministry of natural resources introduced and answered questions about the special action plan for mangrove protection and restoration, (2020-2025)*. Available at: http://www.gov.cn/xinwen/2020-08/31/content_5538800.htm.

Nguyen, H. T. T., Hardy, G. E. S., Van Le, T., Nguyen, H. Q., Huy Nguyen, H., et al. (2021). Mangrove forest landcover changes in coastal Vietnam: A case study from 1973 to 2020 in thanh hoa and nghe an provinces. *Forests* 12 (5), 637.

Pham, T. D., and Yoshino, K. (2016). Impacts of mangrove management systems on mangrove changes in the Northern Coast of Vietnam. *Tropics.* 24, 141–151.

Primavera, J. H., and Esteban, J. M. (2008). A review of mangrove rehabilitation in the Philippines: Successes, failures and future prospects. *Wetlands Ecol. Manage.* 16, 345–358.

Qiu, P., Zhang, P., Rutana, Y. A., and Zhou, X. (2018). *The integration of marine cultural industry and Tourism* (Ocean Development and Management), 35 (04), 16–20. doi: 10.20016/j.cnki.hykfygl.2018.04.003

Rezazadeh, S., Jahani, A., Makhdoum, M., and Meigooni, H. G. (2017). Evaluation of the strategic factors of the management of protected areas using SWOT analysis-case study: Bashgol protected area-qazvin province. *Open J. Ecol.* 07 (1), 55–68. doi: 10.4236/oje.2017.71005

Rong, Y., and Rui, S. (2015). Ecology compensatory mechanism research of extractive industries in bayannur based on SWOT analysis. *Environ. Dev*. 2015 (5), 14–18. doi: 10.16647/j.cnki.cn15-1369/x.2015.05.004

Roslan, Z., Ramli, Z., and Choy, E. A. (2018), 61–67.

Saidul, I. Md. (2014). *Confronting the blue revolution: Industrial aquaculture and sustainability in the global south* (Toronto: University of Toronto Press), 19–148.

Sartor, P., Carbonara, P., and Cerasi, S. (2019). A selective and low impacting traditional fishery, sustaining the economy of small coastal villages in central Mediterranean: Keep or replace the small-scale driftnets. *Fisheries Manage. Ecol.* 26 (6), 661–673. doi: 10.1111/fme.12397

Sayyed, M., Mansoori, M. S., and Jaybhaye, R. G. (2013). SWOT analysis of tandooreh national park (NE Iran) for sustainable ecotourism. *Proc. Int. Acad. Ecol. Environ. Sci.* 3 (4), 296–305.

Sha, M. C., and Zhu, R. J. (2021). "Consideration on temperature control technology of circulating water system in industrialized aquaculture," in *2021 International Conference on Intelligent Manufacturing Technology and Information Technology*. (China) Vol. 11. 90–92.

Suantika, G., Dhert, P., Rombaut, G., Vandenberghe, J., De Wolf, T., and Sorgeloos, P. (2001). The use of ozone in a high-density recirculation system for rotifers. *Aquaculture* 35–49. doi: 10.1016/S0044-8486(01)00532-4

Su, Y. J., Xu, J. J., Marios, S., and Shen, S. W. (2021). Authenticity, perceived value and loyalty in marine tourism destinations: The case of zhoushan, zhejiang province, China. *Sustainability* 13 (7), 21–25. doi: 10.3390/su13073716

Tiller, R. G. (2015). Confronting the blue revolution: Industrial aquaculture and sustainability in the global south. *Global Environ. politics* 15 (3), 186–187. doi: 10.1162/GLEP_r_00319

Wang, W. (2017). Initial analysis on the environmental superiority of "Fisheries solar energy complementary" photovoltaic power plant in northern jiangsu province. *pollut. Control Technol.* 28, 30–33.

Wang, Z. (2020). Development and integration model of marine tourism resources based on sustainable development. *J. Coast. Res.* 103, 1030–1033. doi: 10.2112/SI103-214.1

Xinhuanet (2022) *Focus on high-quality development | mangrove "turning green into gold" to explore a new way of high-quality development*. Available at: https://baijiahao.baidu.com/s?id=1734219295794292261&wfr=spider&for=pc.

Xinhua News Agency (2018) *Notice on strengthening the protection of coastal wetlands and strictly controlling reclamation*. Available at: http://www.gov.cn/xinwen/2018-07/25/content_5309087.htm.

Xinhua News Agency (2021). *The main data results of the third national land survey were released*. Available at: http://www.gov.cn/xinwen/2021-08/26/content_5633497.htm

Xu, F. Y. (2012). Study on development of Sea eco-tourism based on stakeholders 'Analysis. *Resource Dev. Market* 28 (07), 657–659+656.

Xu, S. S., Zhao, D. L., Xie, J. Q., and Zhang, B. X. (2021a). Enlightenment of "Ecological bank" on marine ecological restoration and value realization: Conception of establishing "Mangrove ecological bank". *Ocean Dev. Manage.* 38 (12), 67–73. doi: 10.3969/j.issn.1005-9857.2021.12.010

Xu, S. S., Zheng, Z. Y., Lu, Y., and Zhang, B. X. (2021b). Exploring and solving the difficult problem of marine ecological restoration from the relationship of policy, funds and technology. *Ocean Dev. Management.* 38 (6), 65–69. doi: 10.3969/j.issn.1005-9857.2021.06.011

Yang, C. J. (2002). Discussion on the division of suitable forest land for mangrove forest. *Cent. South For. inventory planting* 21 (2), 2.

Yang, Z., and Guo, J. (2017). A study on the realization path of marine ecological security governance modernization in China. *J. Ocean Univ. China (Social Sciences)* 37, 19–23. doi: 10.16497/j.cnki.1672-335x.2017.06.004

Yu, L. Y., Lin, S. H., Jiao, X. Y., Shen, X. X., and Li, R. (2019). Ecological problems and protection countermeasures of mangrove wetland in guangdong-Hong Kong-Macao greater bay area. *Acta Scientiarum Naturalium Universitatis Pekinensis* 55 (4), 782–790. doi: 10.13209/j.0479-8023.2019.051

Zhang, Q., Zhou, X., Guo, B., Bao, Y., and Wang, Q. (2020). Synergetic realization path of water ecological civilization construction and human-land system optimization under the "Three elements" structure. *J. Coast. Res.*, 104 (SI): 854–857. doi: 10.2112/JCR-SI104-146.1

Zheng, Y. X., and Zhu, Y. (2020). A study on marketing strategy of marine tourism in Yangtze river delta based on SWOT perspective. *J. Innovation Soc. Sci. Res.* 7 (12), 25–28.

frontiers | Frontiers in Marine Science

TYPE Mini Review
PUBLISHED 27 February 2023
DOI 10.3389/fmars.2023.1033258

Check for updates

OPEN ACCESS

EDITED BY
Nathan Knott,
NSW Government, Australia

REVIEWED BY
Eva Aylagas,
King Abdullah University of Science and Technology, Saudi Arabia
Matt Rees,
NSW Government, Australia

*CORRESPONDENCE
Alison K. S. Wee
kimshan.wee@nottingham.edu.my
Tadashi Kajita
kajita@mail.ryudai.jp

SPECIALTY SECTION
This article was submitted to Marine Conservation and Sustainability, a section of the journal Frontiers in Marine Science

RECEIVED 31 August 2022
ACCEPTED 13 February 2023
PUBLISHED 27 February 2023

CITATION
Wee AKS, Salmo III SG, Sivakumar K, Then AY-H, Basyuni M, Fall J, Habib KA, Isowa Y, Leopardas V, Peer N, Artigas-Ramirez MD, Ranawana K, Sivaipram I, Suleiman M and Kajita T (2023) Prospects and challenges of environmental DNA (eDNA) metabarcoding in mangrove restoration in Southeast Asia. *Front. Mar. Sci.* 10:1033258. doi: 10.3389/fmars.2023.1033258

Prospects and challenges of environmental DNA (eDNA) metabarcoding in mangrove restoration in Southeast Asia

Alison K. S. Wee[1*], Severino G. Salmo III[2], Kannan Sivakumar[3], Amy Y-H Then[4], Mohammad Basyuni[5], Jean Fall[6], Kazi Ahsan Habib[7], Yukinobu Isowa[8], Venus Leopardas[9], Nasreen Peer[10], Maria D. Artigas-Ramirez[8], Kithsiri Ranawana[11], Itchika Sivaipram[12], Monica Suleiman[13] and Tadashi Kajita[8,13*]

[1]School of Environmental and Geographical Sciences, University of Nottingham Malaysia Campus, Semenyih, Selangor, Malaysia, [2]Institute of Biology, College of Science, University of the Philippines Diliman, Quezon City, Philippines, [3]Centre of Advanced Study in Marine Biology, Faculty of Marine Sciences, Annamalai University, Portonovo, India, [4]Institute of Biological Sciences, Faculty of Science, Universiti Malaya, Kuala Lumpur, Malaysia, [5]Center of Excellence for Mangrove, Universitas Sumatera Utara, Medan, Indonesia, [6]University Institute of Fisheries and Aquaculture, University Cheikh Anta Diop, Dakar, Senegal, [7]Dept. of Fisheries Biology and Genetics, Faculty of Fisheries, Aquaculture & Marine Science, Sher-e-Bangla Agricultural University, Dhaka, Bangladesh, [8]Iriomote Station, Tropical Biosphere Research Center, University of the Ryukyus, Uehara, Taketomi, Yaeyama, Okinawa, Japan, [9]College of Marine and Allied Sciences, Mindanao State University at Naawan, Naawan, Misamis Oriental, Philippines, [10]Department of Botany and Zoology, Stellenbosch University, Stellenbosch, South Africa, [11]Department of Zoology, Faculty of Science, University of Peradeniya, Peradeniya, Sri Lanka, [12]Department of Marine Science, Faculty of Science, Chulalongkorn University, Bangkok, Thailand, [13]Institute for Tropical Biology and Conservation, Universiti Malaysia Sabah, Kota Kinabalu, Sabah, Malaysia

Species detection using environmental DNA (eDNA) is a biomonitoring tool that can be widely applied to mangrove restoration and management. Compared to traditional surveys that are taxa-specific and time-consuming, eDNA metabarcoding offers a rapid, non-invasive and cost-efficient method for monitoring mangrove biodiversity and characterising the spatio-temporal distribution of multiple taxa simultaneously. General guidelines for eDNA metabarcoding are well-established for aquatic systems, but habitat-specific guidelines are still lacking. Mangrove habitats, as priority ecosystems for restoration in Southeast Asia, present unique prospects and challenges in these regards. Environmental DNA metabarcoding can be used to (1) track functional recovery in ecological restoration, (2) prioritise conservation areas, (3) provide early warning for threats, (4) monitor threatened taxa, (5) monitor response to climate change, and (6) support community-based restoration. However, these potential applications have yet been realized in Southeast Asia due to (1) technical challenges, (2) lack of standardised methods, (3) spatio-temporal difficulties in defining community, (4) data limitations, and (5) lack of funding, infrastructure and technical capacity. Successful implementation of eDNA metabarcoding in mangrove restoration activities would encourage the development of data-driven coastal management and equitable conservation programs. Eventually, this would promote Southeast Asia's shared regional interests in food security, coastal defence and biodiversity conservation.

KEYWORDS

biodiversity, conservation, environmental DNA, metabarcoding, monitoring, rehabilitation

1 Introduction

Mangroves provide various ecosystem services and socio-economic benefits (Hochard et al., 2019; Spalding and Parrett, 2019; Friess et al., 2020). Mangroves are also recognized for their roles in climate change adaptation and mitigation (CCAM; Zari et al., 2019) and biodiversity conservation (da Rosa and Marques, 2022). The largest (32.2%; 43,767 km^2; Spalding and Leal, 2021) and most diverse mangroves (> 50 species; Spalding, 2010) are found in Southeast Asia (SEA). Although mangrove degradation rate declined globally in the last 10 years, SEA remains a hotspot for mangrove loss (Spalding and Leal, 2021; Bhowmik et al., 2022), with Myanmar and the Philippines experiencing the highest rate of loss (Long et al., 2014; Estoque et al., 2018). Aquaculture is the main driver of loss in most SEA countries, though other drivers could be country-specific, e.g. rice farms in Myanmar, and oil palms in Malaysia and Indonesia (Richards & Friess, 2016). When mangroves are lost or degraded, the delivery of ecosystem services and biodiversity conservation are expected to decline (Sannigrahi et al., 2020).

Mangroves and SEA are therefore priority ecosystems and regions, respectively, for global CCAM and biodiversity conservation (Bunting et al., 2022). Mangrove restoration is advocated as a nature-based solution (NbS; Zimmer et al., 2022) and an integral component of the United Nations Decade on Ecosystem Restoration (Su et al., 2021). The region has already implemented several restoration programs since the 1990s (e.g. Walters, 2003; Ilman et al., 2011; Hamdan et al., 2014) although the efficacy was not systematically assessed and reported (Lee et al., 2019). Generally, when mangroves are restored, the vegetation and sediment conditions are expected to improve with age of the stands (Salmo et al., 2013), followed by improvements in the faunal assemblages and associated food web indicators (Salmo et al., 2017; Then et al., 2021; Basyuni et al., 2022). As more researchers move away from simple metrics of forest cover to assess the efficacy of restoration programs, regular monitoring and documentation of biodiversity becomes more important in mangrove restoration ecology (da Rosa and Marques, 2022). In this regard, conventional biodiversity monitoring approaches (e.g. faunal sampling, transect and plot techniques) remain vital to undoubtedly provide foundational empirical data but are time consuming and expensive (Taddeo and Dronova, 2018).

The development of environmental DNA (eDNA) provides an opportunity for biodiversity documentation that can complement conventional biodiversity monitoring techniques (e.g. Oka et al., 2021; Polanco Fernández et al., 2021). The term eDNA refers to organismal DNA and extra-organismal DNA sourced from marine, aquatic, aerial and terrestrial environmental samples, such as water and sediments (Rodriguez-Ezpeleta et al., 2021). Environmental DNA metabarcoding is a novel method of assessing biodiversity from a wide taxonomic group using remnant DNA from environmental samples (Ruppert et al., 2019; see Figure 1 for the workflow of eDNA metabarcoding). The utilisation of eDNA metabarcoding has revolutionised biodiversity research in that (1) it is faster than conventional biodiversity assessment using morphological identification, making it efficient and relatively cheap, (2) the sampling process is simple, non-destructive, and non-invasive, (3) it can detect rare, cryptic, and elusive species, (4) it enables the early detection of imminent, significant environmental change, (5) it is standardised and reproducible across different life stages and environment, and (6) it allows for the simultaneous biodiversity assessment for a wide range of organisms (Beng & Corlett, 2020). Despite its increased popularity in aquatic and marine ecosystems (Bessey et al., 2021), eDNA metabarcoding is still not widely adapted yet in SEA due to several key technical and practical limitations. In this mini-review, we synthesize the viability, potentials, challenges and future prospects of applying eDNA metabarcoding approaches in mangrove restoration in SEA. We align our recommendations to the broader aims of the UN's Decade on Ecosystem Restoration and highlight the role of eDNA metabarcoding in advancing mangrove restoration research in SEA.

2 Environmental DNA applications for mangrove restoration

2.1 Track functional recovery in ecological restoration

The success of mangrove restoration to date is often measured by forest cover recovery using remote sensing and quadrat/transect surveys (Giri, 2016). This is because measuring the recovery of all ecosystem components (and thus the full ecosystem function – Bosire et al., 2008) is more challenging, especially the recovery of mangrove biota such as macrofaunal, meiofaunal and microbial communities. Often, excavation, careful taxonomic identification, and abundance/biomass quantification, are needed to obtain meaningful results. Thus, assessing restoration success can often be time-consuming, resource-intensive, and subjective due to the limited availability of expertise. Environmental DNA metabarcoding would allow for faster, accurate and cost-efficient data collection at a large number of sites, including healthy and degraded ones (Wang et al., 2019; Saenz-Agudelo et al., 2022), and for tracking community-wide shifts in response to recovery (but see quantification limitation in Section 3.4). The possibility to standardise sampling protocols across time and user groups allows for an accurate chronosequence of species succession. Furthermore, the high resolution of eDNA metabarcoding data can improve understanding of fine-scale movement of taxa or connectivity between systems. Environmental DNA also allows for the detection of rare species, and the accurate identification of specimens that would be difficult through morphological examination alone, e.g. cryptic species (Lim et al., 2016; Mennesson et al., 2018) or juvenile/larval stages (Marshall and Stepien, 2019).

2.2 Prioritise conservation area

Evidence from aquatic and marine ecosystems have demonstrated the usefulness of eDNA metabarcoding in informing protected area design. Significant differences in fish species composition among coral reefs with different levels of protection in Lombok Island (Indonesia) was observed using eDNA metabarcoding (Gelis et al., 2021). Environmental DNA techniques can also be used in the conservation management of priority species, especially in monitoring populations and understanding habitat boundaries. For example, in blacktip reef sharks, single-species eDNA surveys using real-time PCR detected spatio-temporal changes in abundance that are comparable to extensive fishing surveys and acoustic telemetry (Postaire et al., 2020), highlighting the potential of eDNA for monitoring shark populations and understanding their habitat boundaries. These eDNA approaches can be applied to the mangrove habitat to understand the movement ecology of priority species and the connectivity across habitats. Such information on metapopulation dynamics will aid in the prioritisation and design of conservation areas and the selection of restoration sites.

2.3 Provide early warning for threats

Environmental DNA metabarcoding allows for the early detection of elusive and invasive species at low densities, which is otherwise challenging to monitor using conventional techniques (Loeza-Quintana et al., 2021). Environmental DNA has been used to monitor the invasive Burmese python *Python bivittatus* in South Florida (Piaggio et al., 2014) and the Atlantic Charru mussel *Mytella strigata* in Singapore (Ip et al., 2021). Also, significant changes in indicator copepod species like *Paracalanus indicus* and *Hexanauplia* detected by eDNA metabarcoding can be used to monitor heat waves and their impacts on aquatic ecosystems (Berry et al., 2019). Interestingly, the recent development of 60 species-specific real-time, or quantitative, PCR (qPCR) assays for invasive, threatened, and exploited freshwater vertebrates and invertebrates in North America (Hernandez et al., 2020) paved the way forward for other aquatic ecosystems, e.g. mangroves, that are constantly under threat. In mangroves, invasive plants (e.g. carrotwood) and harmful algae could cause significant loss of biodiversity and damage to ecosystem function. Thus, early detection and regular monitoring of these species using eDNA metabarcoding and even species-specific qPCR assays is crucial to ensure successful ecological restoration.

2.4 Monitor threatened taxa

Mangroves in SEA are key habitats of multiple threatened species such as the dugong (*Dugong dugon*), proboscis monkey (*Nasalis larvatus*), roughnose stingray (*Pastinachus solocirostris*) and tiger tail seahorse (*Hippocampus comes*). Reliable monitoring of these organisms is crucial for data-driven conservation actions but remains a challenge owing to the lack of standardised methods, elusiveness of the species, and dependence on species experts (Thomsen et al., 2012). Environmental DNA has various proven applications in monitoring specific threatened taxa, including uncovering previously unrecorded elasmobranch species in Singapore's waters (Ip et al., 2021), mapping the distribution of endangered European eel (Weldon et al., 2020) and documenting rare and threatened sharks and rays across eastern Indonesia (Moore et al., 2021). This is accelerated by the development of universal primers that allows for metabarcoding and multi-taxa detection, e.g. the reptile primers simultaneously detected the vulnerable flatback turtle and the Indo-Australian water snake, which inhabit mangrove forests in SEA (West et al., 2021). Furthermore, species detection by eDNA can now include qPCR, whereby the presence and even abundance of a species can be quantitatively estimated based on the eDNA concentration (Weltz et al., 2017). Species detection by qPCR of eDNA samples is still less expensive than traditional surveys, and represent a highly repeatable and sensitive method for behaviorally elusive species (Qu & Stewart, 2019). The presence and recovery of threatened taxa can thus be used as an important metric in monitoring mangrove restoration success.

2.5 Monitor response to climate change

Environmental DNA metabarcoding is useful in tracking the response of mangrove communities to climate change. For example, a 5-year eDNA metabarcoding survey demonstrated a significant seasonal change in meroplankton communities, including fish, molluscs, and cnidarians, especially after the 2015 strong El Niño event (Djurhuus et al., 2020). In 2021, UNESCO launched a global eDNA metabarcoding expedition to study species vulnerability to climate change at marine World Heritage sites (UNESCO, 2021). As successful restoration includes resilience towards climate change, similar eDNA metabarcoding approaches can be applied to mangrove ecosystems to understand climatic response and facilitate adaptation.

2.6 Support community-based restoration

The benefits of involving local communities in restoration efforts and monitoring are three-fold (Schmitt and Duke, 2015; Miya et al., 2022): (1) shared ownership ensures multi-stakeholder support and continued protection of the site, (2) payment for restoration efforts contribute towards local livelihoods, and, (3) increased participation allows for more detailed data collection. However, the lack of taxonomic expertise and the need to maintain a standardised protocol present challenges to increase community involvement in restoration (Eger et al., 2022). The use of eDNA metabarcoding addresses these limitations as community volunteers or employees are able to collect the eDNA metabarcoding sample with minimal training (Miya et al., 2022), and samples are then analysed by taxon experts and molecular researchers who can easily provide a comprehensive species list. This rapid assessment by local communities across a finer temporal and spatial scale (Agersnap et al., 2022) could also allow for early detection of invasive species (Larson et al., 2020) and changes in coastal communities or targeted species (Biggs et al., 2015). In addition, it is important to sustain the collaboration between molecular ecologists and local stakeholders through the Adoption of Translational Molecular Ecology that enhances consensus on objectives, methods, and outcomes of environmental management projects (Aylagas et al., 2020). Establishing a sustained dialogue among stakeholders is key to accelerating the adoption of molecular-based approaches for marine monitoring and assessment (Aylagas et al., 2020).

3 Challenges

3.1 Technical challenges

The eDNA metabarcoding workflow – from sample collection, preservation, amplification, sequencing to bioinformatic analysis – presents multiple technical challenges. As eDNA metabarcoding work involves detecting minute amounts of degraded DNA from environmental samples, the sample collection process is therefore prone to contamination, from sampling instruments, lab supplies (e.g. collection tubes), boats and other field gear (e.g. boots). Therefore, field decontamination before sampling is paramount to ensure sample independence (Goldberg et al., 2016). The most widely-adopted approach to capture eDNA is by filtration, which is highly dependent on factors like pH, amount of suspended particles, filter pore size, and filtered water volume (Majaneva et al., 2018; Wong et al., 2020). For the turbid estuarine waters characteristic of mangroves in SEA, high concentrations of suspended organic and inorganic particles are associated with higher eDNA abundance but also rapid clogging of filters. Sample preservation is key, as the degradation of eDNA accelerated by the warm humid tropical climate results in shorter DNA fragments (Ruppert et al., 2019) that reduce PCR amplification success. Also, studies in relatively remote areas can pose logistical challenges in sample preservation. Higher levels of PCR inhibitions are also linked to turbid waters (Kumar et al., 2022); some inhibitors, such as humic acid, fulvic acid, tannic acid, and hematin, are naturally excreted in the environment from animals or plants (Sato et al., 2017; Minegishi et al., 2019; Uchii et al., 2019). In highly biodiverse tropical mangroves, successful co-amplification of eDNA strongly depends on suitable design primer(s) that is (are) specific and sensitive (Coissac et al., 2012). These challenges would have to be overcome to facilitate the standardisation of experimental protocol.

3.2 Lack of standardised methods

The standardisation of methodology is necessary to ensure comparability across studies. Taxon detection using eDNA metabarcoding is highly dependent on environmental variables (Stewart, 2019; Blabolil et al., 2021). Taxa groups may differ based on the types of waters sampled in the mangroves (Majaneva et al., 2018; Jerde et al., 2019) – eDNA from estuarine waters will likely reflect both surface and bottom estuarine fauna such as fish and prawns but interstitial or pore water may better represent terrestrially-associated fauna, e.g. gastropods. Other sources such as aquatic sediments may contain more eDNA of fish as compared to water samples (Turner et al., 2015). The lack of consensus on best practices for collection and analysis (Goldberg et al., 2015) ultimately prevents the development of a universal eDNA protocol for SE Asian mangroves and reduces the comparability of biodiversity studies (Fonseca, 2018).

3.3 Spatio-temporal difficulties in defining community

The terrestrial, aquatic and intertidal fauna found in mangroves are diverse in life history strategies, and may use mangroves for part of or throughout their lifespans. Mobile taxa such as fish may use mangrove creeks and estuaries exclusively or opportunistically in conjunction with other adjacent habitats, and may move upstream even into more freshwater zones (Krumme, 2004; Russell and McDougall, 2005). Hence, taxon detection depends on multiple factors, e.g. tidal condition, monsoons, salinity gradient, and stratification of the water column (Figure 2). Furthermore, understanding processes that govern both eDNA release from focal mangrove fauna and removal of eDNA in the intertidal mangrove environment, i.e. eDNA sources and sinks respectively, are critical (Stewart, 2019) but remain poorly documented. The dynamic spatio-temporal heterogeneity in faunal utilisation of mangroves also poses a problem of scale in terms of delineating the sampling boundaries for different ecological and ecosystem applications. Due to the bio-geomorphological complexity of mangrove habitats, the species distribution data could be influenced by the microhabitat, estuarine position and extent, biogeographic region and mangrove type (Figure 2), in addition to being confounded by human influences (e.g. the presence of introduced or invasive species). Hence, it is important that eDNA metabarcoding in mangroves includes sampling at multiple spatial scales and checklist records of native species to allow for better understanding of species distribution. Consensus in sampling protocols at congruent scales is needed to facilitate comparative studies across mangroves in SEA.

3.4 Data limitations

A comprehensive reference database is lacking for many highly speciose but poorly studied mangrove fauna (Rajpar and Zakaria, 2014; Jerde et al., 2021). Across all taxa, fishes (especially freshwater fish) have the most comprehensive reference database, e.g. FISH-BOL (Ward et al., 2009). However, the available reference databases are mostly region-specific and have limited transferability across ecosystems and ecoregions of the world. The absence of a region-specific reference database has confined eDNA metabarcoding research in SEA to single-species study on ecologically significant taxa, e.g. highly invasive bivalve (Xia et al., 2018). As species identification using eDNA metabarcoding is only as good as the reference database (Thomsen & Sigsgaard, 2019), establishing a regional, comprehensive database is the necessary step forward for the application of eDNA metabarcoding as a routine biomonitoring tool.

Environmental DNA metabarcoding holds great promise in determining relative species abundance and/or biomass. To date, abundance/or biomass estimation has been shown feasible for select taxa in 'controlled' environments such as aquaria and in selected natural environments (see review by Rourke et al., 2022). For this to be feasible, correlations must be established between eDNA concentrations and abundance and/or biomass across taxa (Fonseca, 2018). Further work will be essential to overcome eDNA data limitations beyond documentation of species presence.

3.5 Lack of funding, infrastructure and capacity building

Although eDNA metabarcoding can be more cost-effective than traditional approaches in highly biodiverse regions (Bálint et al., 2018), it remains relatively cost-prohibitive in local SEA currencies, especially when many molecular supplies come from developed countries like the USA and Germany. Advances in eDNA biodiversity applications to date are primarily confined to high-income developed countries (Rourke et al., 2022). This is further hampered by limitations in funding, molecular laboratories and infrastructure that can support the post-sampling analyses. The molecular component poses a steep learning curve, and coupled with the lack of user-friendly reference databases, hampers the uptake of eDNA metabarcoding within SEA. Knowledge transfer and capacity building training that promote hands-on experiences

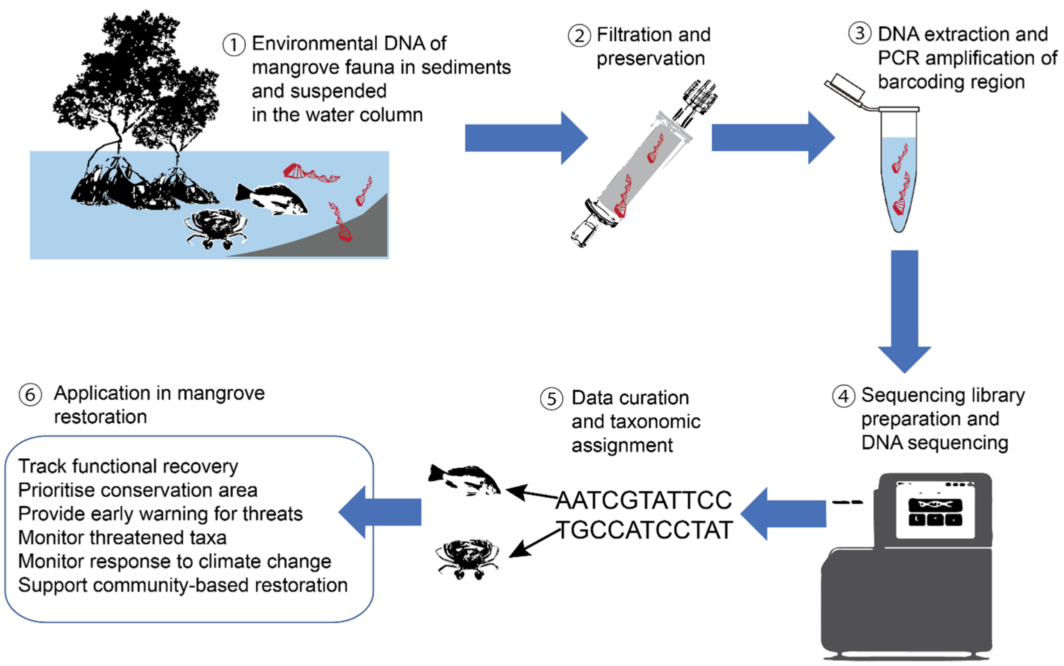


FIGURE 1
Workflow and potential applications of eDNA metabarcoding in mangroves.

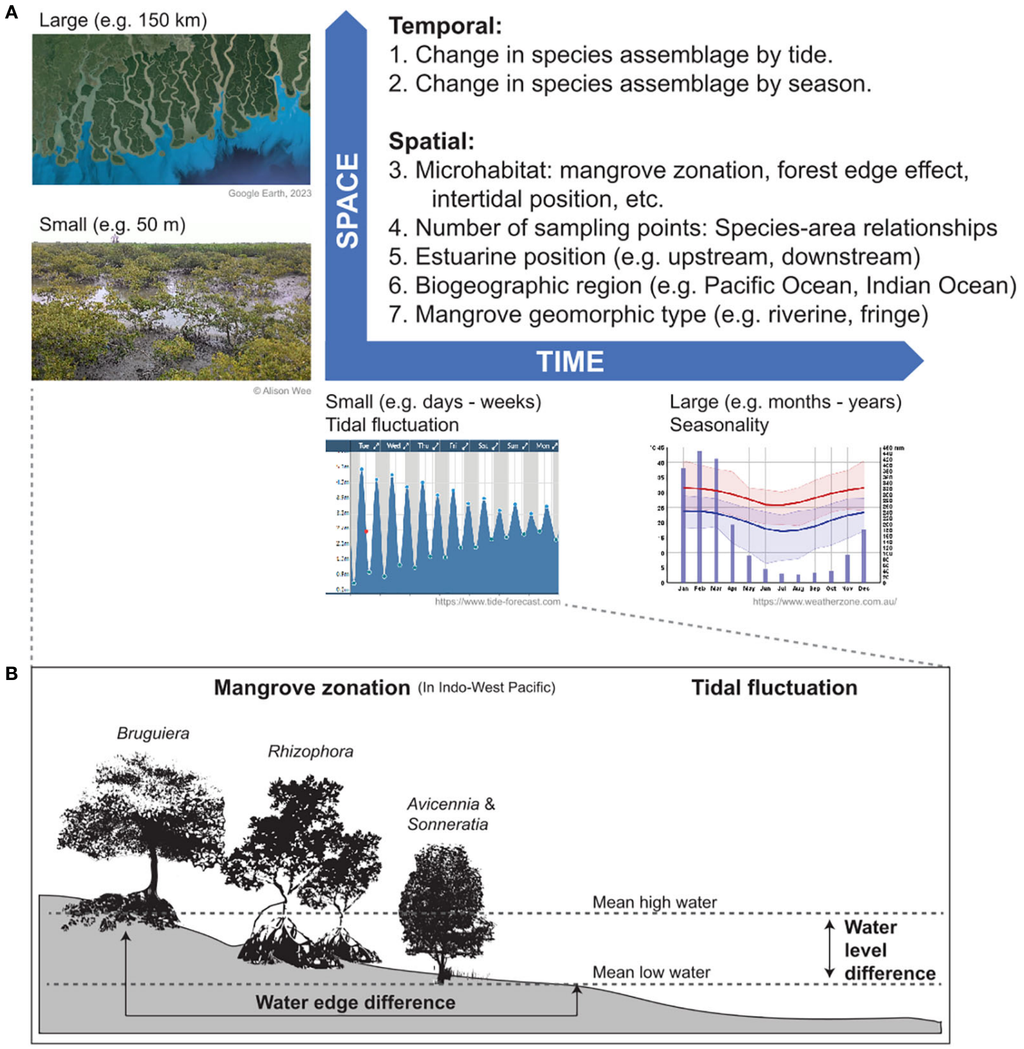


FIGURE 2
(A) Key examples of spatio-temporal challenges of designing eDNA metabarcoding studies in mangroves, **(B)** mangrove zonation and tidal fluctuation are two examples of the spatio-temporal challenges in mangrove eDNA metabarcoding study design.

in field sampling, laboratory and data analysis, as well as mutual academic exchanges to develop the best practice, are necessary.

4 Future perspectives

For eDNA metabarcoding to fully realise its potential as a biomonitoring tool in mangrove restoration in SEA, several key organisational, technical and logistical improvements will need to be implemented. First, cohesive international and regional collaboration is necessary to overcome the data limitations, enable capacity building and facilitate the much-needed development of a comprehensive reference database. Successful endeavours have been undertaken at the national level, e.g. the Atlas of Living Australia (hosted by CSIRO) and the Biodiversity of Singapore (hosted by the Lee Kong Chian Natural History Museum) initiatives, whereby concerted efforts among government agencies, scientists and citizens have led to the establishment of a well-curated biodiversity inventory supplemented by DNA barcoding. To advance eDNA metabarcoding in mangrove habitats with pantropical distribution, transboundary cooperation is essential, and hence such efforts should be emulated at an international level. International collaboration will benefit not only the development of the reference database, but also encourages technical exchanges and the sharing of facilities and resources. All of this will be crucial for developing countries in SEA and elsewhere, which are commonly underfunded for molecular biodiversity research. Long term funding, preferably from multiple sources, would ensure the continuity of the collaborative platform. One key example is the "Global analyses of mangrove ecosystem by eDNA metabarcoding" international research project supported by the Japan Society for the Promotion of Science (JSPS) Core-to-Core, which enabled capacity-building through seminars and workshops and eventually led to spin-off projects funded by regional, multinational sources. One of the spin-offs being the international collaborative research project between Japan, Philippines, and

Indonesia on the "Application of eDNA metabarcoding in faunal biodiversity assessment of Indo-Pacific mangroves vulnerable to climate change" under the East Asia Science and Innovation Area Joint Research Program (e-ASIA JRP), supported by public co-funding from the East Asia Summit (EAS) member countries.

Second, explorative studies are essential in standardising the methodology, developing the collaborative platform, and building the baseline data for biomonitoring. Regular meetings among the regional mangrove eDNA metabarcoding community are needed to coordinate and refine a common methodology for local use. A standardised, freely available experimental manual, e.g. the Environmental DNA Sampling and Experiment Manual (The eDNA Society, 2019), would be an ideal starting point, upon which technical improvements to overcome excessive sediments in samples and PCR inhibitors can be based. To address the difficulties in defining the sampling scale, it is important to first conduct baseline studies that sample widely and across various spatio-temporal and forest configurations. This would include sampling across tidal regimes, monsoonal seasons, intertidal zones, water depths, salinity gradients, mangrove habitat types (e.g. fringe, estuarine), spatial scales and levels of anthropogenic impacts. Following this, a collaborative platform for inventorizing the eDNA samples would be necessary to facilitate the comparison of biodiversity across spatio-temporal configurations. For example, the ANEMONE DB (https://db.anemone.bio/) based in Tohoku University provides targeted inventorizing of MiFish eDNA metabarcoding surveys in the Pacific Ocean and the WilderLab platform (https://www.wilderlab.co.nz/) showcased publicly available eDNA data in New Zealand. Such a data repository would facilitate the sharing of eDNA data that can eventually inform decision-making on sampling design and workload planning. In addition, the initial studies would provide essential baseline data for future biomonitoring efforts.

In summary, this review provides an essential, practical guide to scientists, policymakers, conservation practitioners and mangrove forest managers in implementing eDNA metabarcoding as a biomonitoring tool in mangrove restoration programs. The implementation of eDNA metabarcoding would encourage the development of data-driven coastal management and equitable conservation programs. Such advancement is especially needed in SEA as it comprises coastal nations with shared coastal resources, threats and ecosystem restoration goals. Hence, international collaboration and capacity building in mangrove eDNA metabarcoding are crucial to promote the region's interests in food security, coastal defense and biodiversity conservation.

Author contributions

All authors contributed to the conceptualization, literature review and writing. All authors contributed to the article and approved the submitted version.

Funding

This study is an output from the 2nd International Workshop for Mangrove Biodiversity Studies by eDNA Metabarcoding supported by the Japan Society for the Promotion of Science (JSPS) Core-to-Core Program (JPJSCCB20200007). The work is also supported by the Japan Science and Technology Agency (JST) SICORP (e-ASIA JPMJSC21E5 and AJ-CORE JPMJSC21E5), and the JSPS KAKENHI Fund for the Promotion of Joint International Research (21KK0115). AW, AT and TK are supported by the Fundamental Research Grant Scheme FRGS 2021 (FRGS/1/2021/WAB11/UNIM/02/1) awarded by the Ministry of Higher Education Malaysia. AW, AT, MB and TK are supported by the Collaborative Regional Research Programme 2020 (CRRP2020-SP1103-Wee) awarded by the Asia-Pacific Network for Global Change Research. SS was supported by the Foreign Researcher Invitation Program at the University of the Ryukyus. MB was supported by Basic Research Collaboration Scheme 2022-2024 (e-Asia Joint Research Program) (079/UN5.2.3.1/PPM/KP-DRTPM/L/2022) from the Ministry of Education, Culture, Research and Technology of Government of Indonesia.

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.

Publisher's note

References

Agersnap, S., Sigsgaard, E. E., Jensen, M. R., Avila, M. D. P., Carl, H., Møller, P. R., et al. (2022). A national scale "BioBlitz" using citizen science and eDNA metabarcoding for monitoring coastal marine fish. *Front. Mar. Sci.* 9. doi: 10.3389/fmars.2022.824100

Aylagas, E., Borja, A., Pochon, X., Zaiko, A., Keeley, N., Bruce, K., et al. (2020). Translational molecular ecology in practice: Linking DNA-based methods to actionable marine environmental management. *Sci. Total Environ.* 744, 140780. doi: 10.1016/j.scitotenv.2020.140780

Bálint, M., Nowak, C., Márton, O., Pauls, S. U., Wittwer, C., Aramayo, J. L., et al. (2018). Accuracy, limitations and cost efficiency of eDNA-based community survey in tropical frogs. *Mol. Ecol. Resour.* 18, 1415–1426. doi: 10.1111/1755-0998.12934

Basyuni, M., Bimantara, Y., Cuc, N. T., Balke, T., and Vovides, A. G. (2022). Macrozoobenthic community assemblage as key indicator for mangrove restoration success in north Sumatra and aceh, Indonesia. *Restor. Ecol.* 30, e13614. doi: 10.1111/rec.13614

Beng, K. C., and Corlett, R. T. (2020). Applications of environmental DNA (eDNA) in ecology and conservation: Opportunities, challenges and prospects. *Biodivers. Conserv.* 29, 2089–2121. doi: 10.1007/s10531-020-01980-0

Berry, T. E., Saunders, B. J., Coghlan, M. L., Stat, M., Jarman, S., Richardson, A. J., et al. (2019). Marine environmental DNA biomonitoring reveals seasonal patterns in biodiversity and identifies ecosystem responses to anomalous climatic events. *PLos Genet.* 15, e1007943. doi: 10.1371/journal.pgen.1007943

Bessey, C., Neil Jarman, S., Simpson, T., Miller, H., Stewart, T., Kenneth Keesing, J., et al. (2021). Passive eDNA collection enhances aquatic biodiversity analysis. *Commun. Biol.* 4, 236. doi: 10.1038/s42003-021-01760-8

Bhowmik, A. K., Padmanaban, R., Cabral, P., and Romeiras, M. M. (2022). Global mangrove deforestation and its interacting social-ecological drivers: A systematic review and synthesis. *Sustainability* 14 (8), 4433. doi: 10.3390/su14084433

Biggs, J., Ewald, N., Valentini, A., Gaboriaud, C., Dejean, T., Griffiths, R. A., et al. (2015). Using eDNA to develop a national citizen science-based monitoring programme for the great crested newt (*Triturus cristatus*). *Biol. Conserv.* 183, 19–28. doi: 10.1016/j.biocon.2014.11.029

Blabolil, P., Harper, L. R., Říčanová, Š., Sellers, G., Di Muri, C., Jůza, T., et al. (2021). Environmental DNA metabarcoding uncovers environmental correlates of fish communities in spatially heterogeneous freshwater habitats. *Ecol. Indic.* 126, 107698. doi: 10.1016/j.ecolind.2021.107698

Bosire, J. O., Dahdouh-Guebas, F., Walton, M., Crona, B. I., Lewis Iii, R. R., Field, C., et al. (2008). Functionality of restored mangroves: a review. *Aquat. Bot.* 89, 251–259. doi: 10.1016/j.aquabot.2008.03.010

Bunting, P., Rosenqvist, A., Hilarides, L., Lucas, R. M., and Thomas, N. (2022). Global mangrove watch: Updated 2010 mangrove forest extent (v2. 5). *Remote Sens.* 14, 1034. doi: 10.3390/rs14041034

Coissac, E., Riaz, T., and Puillandre, N. (2012). Bioinformatic challenges for DNA metabarcoding of plants and animals. *Mol. Ecol.* 21, 1834–1847. doi: 10.1111/j.1365-294X.2012.05550.x

da Rosa, C. M., and Marques, M. C. M. (2022). How are biodiversity and carbon stock recovered during tropical forest restoration? supporting the ecological paradigms and political context involved. *J. Nat. Conserv.* 65, 126115. doi: 10.1016/j.jnc.2021.126115

Djurhuus, A., Closek, C. J., Kelly, R. P., Pitz, K. J., Michisaki, R. P., Starks, H. A., et al. (2020). Environmental DNA reveals seasonal shifts and potential interactions in a marine community. *Nat. Commun.* 11, 254. doi: 10.1038/s41467-019-14105-1

Eger, A. M., Earp, H. S., Friedman, K., Gatt, Y., Hagger, V., Hancock, B., et al. (2022). The need, opportunities, and challenges for creating a standardized framework for marine restoration monitoring and reporting. *Biol. Conserv.* 266, 109429. doi: 10.1016/j.biocon.2021.109429

Estoque, R. C., Myint, S. W., Wang, C., Ishtiaque, A., Aung, T. T., Emerton, L., et al. (2018). Assessing environmental impacts and change in myanmar's mangrove ecosystem service value due to deforestation, (2000–2014). *Glob. Change Biol.* 24, 5391–5410. doi: 10.1111/gcb.14409

Fonseca, V. G. (2018). Pitfalls in relative abundance estimation using eDNA metabarcoding. *Mol. Ecol. Resour.* 18, 923–926. doi: 10.1111/1755-0998.12902

Friess, D. A., Yando, E. S., Wong, L. W., and Bhatia, N. (2020). Indicators of scientific value: An under-recognised ecosystem service of coastal and marine habitats. *Ecol. Indic.* 113, 106255. doi: 10.1016/j.ecolind.2020.106255

Gelis, E. R. E., Kamal, M. M., Subhan, B., Bachtiar, I., Sani, L. M. I., and Madduppa, H. (2021). Environmental biomonitoring of reef fish community structure with eDNA metabarcoding in the coral triangle. *Environ. Biol. Fishes* 104, 887–903. doi: 10.1007/s10641-021-01118-3

Giri, C. (2016). Observation and monitoring of mangrove forests using remote sensing: Opportunities and challenges. *Remote Sens.* 8, 783. doi: 10.3390/rs8090783

Goldberg, C. S., Strickler, K. M., and Pilliod, D. S. (2015). Moving environmental DNA methods from concept to practice for monitoring aquatic macroorganisms. *Biol. Conserv.* 183, 1–3. doi: 10.1016/j.biocon.2014.11.040

Goldberg, C. S., Turner, C. R., Deiner, K., Klymus, K. E., Thomsen, P. F., Murphy, M. A., et al. (2016). Critical considerations for the application of environmental DNA methods to detect aquatic species. *Methods Ecol. Evolution.* 7, 1299–1307. doi: 10.1111/2041-210X.12595

Hamdan, O., Aziz, H. K., and Hasmadi, I. M. (2014). L-band ALOS PALSAR for biomass estimation of matang mangroves, Malaysia. *Remote Sens. Environ.* 155, 69–78. doi: 10.1016/j.rse.2014.04.029

Hernandez, C., Bougas, B., Perreault-Payette, A., Simard, A., Côté, G., and Bernatchez, L. (2020). 60 specific eDNA qPCR assays to detect invasive, threatened, and exploited freshwater vertebrates and invertebrates in Eastern Canada. *Environ. DNA* 2 (3), 373–386. doi: 10.1002/edn3.89

Hochard, J. P., Hamilton, S., and Barbier, E. B. (2019). Mangroves shelter coastal economic activity from cyclones. *Proc. Natl. Acad. Sci. USA* 116, 12232–12237. doi: 10.1073/pnas.1820067116

Ilman, M., Wibisono, I. T. C., and Suryadiputra, I. N. N. (2011). State of the art information on mangrove ecosystems in Indonesia. *Wetlands International-Indonesia Programme Bogor*. 58. Available at: https://indonesia.wetlands.org/ (accessed 20 Feb 2023).

Ip, Y. C. A., Tay, Y. C., Chang, J. J. M., Ang, H. P., Tun, K. P. P., Chou, L. M., et al. (2021). Seeking life in sedimented waters: Environmental DNA from diverse habitat types reveals ecologically significant species in a tropical marine environment. *Environ. DNA* 3, 654–668. doi: 10.1002/edn3.162

Jerde, C. L., Mahon, A. R., Campbell, T., McElroy, M. E., Pin, K., Childress, J. N., et al. (2021). Are genetic reference libraries sufficient for environmental DNA metabarcoding of Mekong river basin fish? *Water* 13, 1767. doi: 10.3390/w13131767

Jerde, C. L., Wilson, E. A., and Dressler, T. L. (2019). Measuring global fish species richness with eDNA metabarcoding. *Mol. Ecol. Resour.* 19, 19–22. doi: 10.1111/1755-0998.12929

Krumme, U. (2004). Patterns in tidal migration of fish in a Brazilian mangrove channel as revealed by a split-beam echosounder. *Fish. Res.* 70, 1–15. doi: 10.1016/j.fishres.2004.07.004

Kumar, G., Farrell, E., Reaume, A. M., Eble, J. A., and Gaither, M. R. (2022). One size does not fit all: Tuning eDNA protocols for high-and low-turbidity water sampling. *Environ. DNA* 4, 167–180. doi: 10.1002/edn3.235

Larson, E. R., Graham, B. M., Achury, R., Coon, J. J., Daniels, M. K., Gambrell, D. K., et al. (2020). From eDNA to citizen science: Emerging tools for the early detection of invasive species. *Front. Ecol. Environ.* 18, 194–202. doi: 10.1002/fee.2162

Lee, S. Y., Hamilton, S., Barbier, E. B., Primavera, J., and Lewis, R. R. (2019). Better restoration policies are needed to conserve mangrove ecosystems. *Nat. Ecol. Evol.* 3, 870–872. doi: 10.1038/s41559-019-0861-y

Lim, H., Zainal Abidin, M., Pulungan, C. P., de Bruyn, M., and Mohd Nor, S. A. (2016). DNA Barcoding reveals high cryptic diversity of the freshwater halfbeak genus *Hemirhamphodon* from sundaland. *PLos One* 11, e0163596. doi: 10.1371/journal.pone.0163596

Loeza-Quintana, T., Crookes, S., Li, P. Y., Reid, D. P., Smith, M., and Hanner, R. H. (2021). Environmental DNA detection of endangered and invasive species in kejimkujik national park and historic site. *Genome* 64, 172–180. doi: 10.1139/gen-2020-0042

Long, J., Napton, D., Giri, C., and Graesser, J. (2014). A mapping and monitoring assessment of the philippines' mangrove forests from 1990 to 2010. *J. Coast. Res.* 30, 260–271. doi: 10.2112/JCOASTRES-D-13-00057.1

Majaneva, M., Diserud, O. H., Eagle, S. H., Boström, E., Hajibabaei, M., and Ekrem, T. (2018). Environmental DNA filtration techniques affect recovered biodiversity. *Sci. Rep.* 8, 4682. doi: 10.1038/s41598-018-23052-8

Marshall, N. T., and Stepien, C. A. (2019). Invasion genetics from eDNA and thousands of larvae: A targeted metabarcoding assay that distinguishes species and population variation of zebra and quagga mussels. *Ecol. Evol.* 9, 3515–3538. doi: 10.1002/ece3.4985

Mennesson, M. I., Bonillo, C., Feunteun, E., and Keith, P. (2018). Phylogeography of eleotris fusca (Teleostei: Gobioidei: Eleotridae) in the indo-pacific area reveals a cryptic species in the Indian ocean. *Conserv. Genet.* 19, 1025–1038. doi: 10.1007/s10592-018-1063-x

Minegishi, Y., Wong, M. K. S., Kanbe, T., Araki, H., Kashiwabara, T., Ijichi, M., et al. (2019). Spatiotemporal distribution of juvenile chum salmon in otsuchi bay, iwate, Japan, inferred from environmental DNA. *PLos One* 14, e0222052. doi: 10.1371/journal.pone.0222052

Miya, M., Sado, T., Oka, S. I., and Fukuchi, T. (2022). The use of citizen science in fish eDNA metabarcoding for evaluating regional biodiversity in a coastal marine region: A pilot study. *Metabarcoding Metagenom.* 6, e80444. doi: 10.3897/mbmg.6.80444

Moore, A. M., Jompa, J., Tassakka, A. C. M. A. R., Yasir, I., Ndobe, S., Umar, W., et al. (2021). Sharks and rays (Chondrichthyes) around Banggai Island, Banggai MPA indonesia: biodiversity data from an environmental DNA pilot study. *AACL Bioflux* 14, 725–745.

Oka, S. I., Doi, H., Miyamoto, K., Hanahara, N., Sado, T., and Miya, M. (2021). Environmental DNA metabarcoding for biodiversity monitoring of a highly diverse tropical fish community in a coral reef lagoon: Estimation of species richness and detection of habitat segregation. *Environ. DNA* 3, 55–69. doi: 10.1002/edn3.132

Piaggio, A. J., Engeman, R. M., Hopken, M. W., Humphrey, J. S., Keacher, K. L., Bruce, W. E., et al. (2014). Detecting an elusive invasive species: A diagnostic PCR to detect b urmese python in f lorida waters and an assessment of persistence of environmental DNA. *Mol. Ecol. Resour.* 14, 374–380. doi: 10.1111/1755-0998.12180

Polanco Fernández, A., Marques, V., Fopp, F., Juhel, J. B., Borrero-Pérez, G. H., Cheutin, M. C., et al. (2021). Comparing environmental DNA metabarcoding and underwater visual census to monitor tropical reef fishes. *Environ. DNA* 3, 142–156. doi: 10.1002/edn3.140

Postaire, B. D., Bakker, J., Gardiner, J., Wiley, T. R., and Chapman, D. D. (2020). Environmental DNA detection tracks established seasonal occurrence of blacktip sharks (*Carcharhinus limbatus*) in a semi-enclosed subtropical bay. *Sci. Rep.* 10, 11847. doi: 10.1038/s41598-020-68843-0

Qu, C., and Stewart, K. A. (2019). Evaluating monitoring options for conservation: comparing traditional and environmental DNA tools for a critically endangered mammal. *Sci. Nat.* 106, 9. doi: 10.1007/s00114-019-1605-1

Rajpar, M. N., and Zakaria, M. (2014). "Mangrove fauna of Asia," in *Mangrove ecosystems of Asia*. Eds. I. Faridah-Hanum, A. Latiff, K. R. Hakeem and M. Ozturk (New York, NY: Springer), 153–197. doi: 10.1007/978-1-4614-8582-7_8

Richards, D. R., and Friess, D. A. (2016). Rates and drivers of mangrove deforestation in southeast asia 2000–2012. *Proc. Natl. Acad. Sci. U.S.A.* 113, 344–349. doi: 10.1073/pnas.1510272113

Rodriguez-Ezpeleta, N., Morissette, O., Bean, C. W., Manu, S., Banerjee, P., Lacoursière-Roussel, A., et al. (2021). Trade-offs between reducing complex terminology and producing accurate interpretations from environmental DNA: Comment on "Environmental DNA: What's behind the term?" by pawlowski et al 2020. *Mol. Ecol.* 30 (19), 4601–4605. doi: 10.1111/mec.15942

Rourke, M. L., Fowler, A. M., Hughes, J. M., Broadhurst, M. K., DiBattista, J. D., Fielder, S., et al. (2022). Environmental DNA (eDNA) as a tool for assessing fish biomass: A review of approaches and future considerations for resource surveys. *Environ. DNA* 4, 9–33. doi: 10.1002/edn3.185

Ruppert, K. M., Kline, R. J., and Rahman, M. S. (2019). Past, present, and future perspectives of environmental DNA (eDNA) metabarcoding: A systematic review in methods, monitoring, and applications of global eDNA. *Glob. Ecol. Conserv.* 17, e00547. doi: 10.1016/j.gecco.2019.e00547

Russell, D. J., and McDougall, A. J. (2005). Movement and juvenile recruitment of mangrove jack, *Lutjanus argentimaculatus* (Forsskål), in northern Australia. *Mar. Freshw. Res.* 56, 465–475. doi: 10.1071/MF04222

Saenz-Agudelo, P., Delrieu-Trottin, E., DiBattista, J. D., Martínez-Rincon, D., Morales-González, S., Pontigo, F., et al. (2022). Monitoring vertebrate biodiversity of a protected coastal wetland using eDNA metabarcoding. *Environ. DNA* 4, 77–92. doi: 10.1002/edn3.200

Salmo, S. G., Lovelock, C., and Duke, N. C. (2013). Vegetation and soil characteristics as indicators of restoration trajectories in restored mangroves. *Hydrobiologia* 720, 1–18. doi: 10.1007/s10750-013-1617-3

Salmo, S. G., Tibbetts, I., and Duke, N. C. (2017). Colonization and shift of mollusc assemblages as a restoration indicator in planted mangroves in the Philippines. *Biodivers. Conserv.* 26, 865–881. doi: 10.1007/s10531-016-1276-6

Sannigrahi, S., Zhang, Q., Pilla, F., Joshi, P. K., Basu, B., Keesstra, S., et al. (2020). Responses of ecosystem services to natural and anthropogenic forcings: A spatial regression based assessment in the world's largest mangrove ecosystem. *Sci. Total. Environ.* 715, 137004. doi: 10.1016/j.scitotenv.2020.137004

Sato, H., Sogo, Y., and Yamanaka, H. (2017). Usefulness and limitations of sample pooling for environmental DNA metabarcoding of freshwater fish communities. *Sci. Rep.* 7, 14860. doi: 10.1038/s41598-017-14978-6

Schmitt, K., and Duke, N. C. (2015). "Mangrove management, assessment and monitoring," in *Tropical forestry handbook*. Eds. M. Köhl and L. Pancel (Berlin, Heidelberg: Springer), 1–29. doi: 10.1007/978-3-642-41554-8_126-1

Spalding, M. (2010). *World atlas of mangroves* (London: Routledge). doi: 10.4324/9781849776608

Spalding, M. D., and Leal, M. (2021). The state of the world's mangroves 2021. *Global Mangrove Alliance*. 79. Available online at: https://www.mangrovealliance.org/mangrove-forests/ (accessed 20 Feb 2023).

Spalding, M., and Parrett, C. L. (2019). Global patterns in mangrove recreation and tourism. *Mar. Policy* 110, 103540. doi: 10.1016/j.marpol.2019.103540

Stewart, K. A. (2019). Understanding the effects of biotic and abiotic factors on sources of aquatic environmental DNA. *Biodivers. Conserv.* 28, 983–1001. doi: 10.1007/s10531-019-01709-8

Su, J., Friess, D. A., and Gasparatos, A. (2021). A meta-analysis of the ecological and economic outcomes of mangrove restoration. *Nat. Commun.* 12, 5050. doi: 10.1038/s41467-021-25349-1

Taddeo, S., and Dronova, I. (2018). Indicators of vegetation development in restored wetlands. *Ecol. Indic.* 94, 454–467. doi: 10.1016/j.ecolind.2018.07.010

The eDNA Society (2019) *Environmental DNA sampling and experiment manual version 2.1*. Available at: http://ednasociety.org/eDNA_manual_Eng_v2_1_3b.pdf (Accessed August 30, 2022).

Then, A. Y. H., Adame, M. F., Fry, B., Chong, V. C., Riekenberg, P. M., Mohammad Zakaria, R., et al. (2021). Stable isotopes clearly track mangrove inputs and food web changes along a reforestation gradient. *Ecosystems* 24, 939–954. doi: 10.1007/s10021-020-00561-0

Thomsen, P. F., Kielgast, J. O. S., Iversen, L. L., Wiuf, C., Rasmussen, M., Gilbert, M. T. P., et al. (2012). Monitoring endangered freshwater biodiversity using environmental DNA. *Mol. Ecol.* 21, 2565–2573. doi: 10.1111/j.1365-294X.2011.05418.x

Thomsen, P. F., and Sigsgaard, E. E. (2019). Environmental DNA metabarcoding of wild flowers reveals diverse communities of terrestrial arthropods. *Ecol. Evol.* 9, 1665–1679. doi: 10.1002/ece3.4809

Turner, C. R., Uy, K. L., and Everhart, R. C. (2015). Fish environmental DNA is more concentrated in aquatic sediments than surface water. *Biol. Conserv.* 183, 93–102. doi: 10.1016/j.biocon.2014.11.017

Uchii, K., Doi, H., Okahashi, T., Katano, I., Yamanaka, H., Sakata, M. K., et al. (2019). Comparison of inhibition resistance among PCR reagents for detection and quantification of environmental DNA. *Environ. DNA* 1, 359–367. doi: 10.1002/edn3.37

UNESCO (2021) *UNESCO Launches global eDNA project to study vulnerability of species to climate change at marine world heritage sites*. Available at: https://ioc.unesco.org/news/unesco-launches-global-edna-project-study-vulnerability-species-climate-change-marine-world (Accessed August 30, 2022).

Walters, B. B. (2003). People and mangroves in the Philippines: fifty years of coastal environmental change. *Environ. Conserv.* 30, 293–303. doi: 10.1017/S0376892903000298

Wang, P., Yan, Z., Yang, S., Wang, S., Zheng, X., Fan, J., et al. (2019). Environmental DNA: an emerging tool in ecological assessment. *Bull. Environ. Contam. Toxicol.* 103, 651–656. doi: 10.1007/s00128-019-02720-z

Ward, R. D., Hanner, R., and Hebert, P. D. (2009). The campaign to DNA barcode all fishes, FISH-BOL. *J. fish Biol.* 74 (2), 329–356. doi: 10.1111/j.1095-8649.2008.02080.x

Weldon, L., O'Leary, C., Steer, M., Newton, L., Macdonald, H., and Sargeant, S. L. (2020). A comparison of European eel *Anguilla anguilla* eDNA concentrations to fyke net catches in five Irish lakes. *Environ. DNA* 2, 587–600. doi: 10.1002/edn3.91

Weltz, K., Lyle, J. M., Ovenden, J., Morgan, J. A. T., Moreno, D. A., and Semmens, J. M. (2017). Application of environmental DNA to detect an endangered marine skate species in the wild. *PLos One* 12 (6), e0178124. doi: 10.1371/journal.pone.0178124

West, K. M., Heydenrych, M., Lines, R., Tucker, T., Fossette, S., Whiting, S., et al. (2021). Development of a 16S metabarcoding assay for the environmental DNA (eDNA) detection of aquatic reptiles across northern Australia. *Mar. Freshw. Re.* doi: 10.1071/MF20288

Wong, M. K. S., Nakao, M., and Hyodo, S. (2020). Field application of an improved protocol for environmental DNA extraction, purification, and measurement using sterivex filter. *Sci. Rep.* 10, 21531. doi: 10.1038/s41598-020-77304-7

Xia, Z., Zhan, A., Gao, Y., Zhang, L., Haffner, G. D., and MacIsaac, H. J. (2018). Early detection of a highly invasive bivalve based on environmental DNA (eDNA). *Biol. Invasions* 20, 437–447. doi: 10.1007/s10530-017-1545-7

Zari, M. P., Kiddle, G. L., Blaschke, P., Gawler, S., and Loubser, D. (2019). Utilising nature-based solutions to increase resilience in pacific ocean cities. *Ecosyst. Serv.* 38, 100968. doi: 10.1016/j.ecoser.2019.100968

Zimmer, M., Ajonina, G. N., Amir, A. A., Crgagg, S. M., Crooks, S., Dahdouh-Guebas, F., et al. (2022). When nature needs a helping hand: Different levels of human intervention for mangrove (re-) establishment. *Front. For. Glob. Change.* 5. doi: 10.3389/ffgc.2022.784322

frontiers | Frontiers in Marine Science

TYPE Original Research
PUBLISHED 23 June 2023
DOI 10.3389/fmars.2023.1127720

OPEN ACCESS

EDITED BY
Severino III Salmo,
University of the Philippines Diliman,
Philippines

REVIEWED BY
Armando Apan,
University of Southern Queensland,
Australia
Ariel Blanco,
University of the Philippines Diliman,
Philippines

*CORRESPONDENCE
Priscilla Baltezar
priscilla.baltezar@gmail.com

RECEIVED 19 December 2022
ACCEPTED 13 April 2023
PUBLISHED 23 June 2023

CITATION
Baltezar P, Murillo-Sandoval PJ, Cavanaugh KC, Doughty C, Lagomasino D, Tieng T, Simard M and Fatoyinbo T (2023) A regional map of mangrove extent for Myanmar, Thailand, and Cambodia shows losses of 44% by 1996.
Front. Mar. Sci. 10:1127720.
doi: 10.3389/fmars.2023.1127720

A regional map of mangrove extent for Myanmar, Thailand, and Cambodia shows losses of 44% by 1996

Priscilla Baltezar[1*], Paulo J. Murillo-Sandoval[2], Kyle C. Cavanaugh[1], Cheryl Doughty[3,4], David Lagomasino[5], Thida Tieng[6], Marc Simard[7] and Temilola Fatoyinbo[3]

[1]Department of Geography, University of California, Los Angeles, Los Angeles, CA, United States, [2]Facultad de Ciencias del Hábitat, Diseño e Infraestructura, University of Tolima, Ibagué, Colombia, [3]Biospheric Sciences Laboratory, National Aeronautics and Space Administration Goddard Space Flight Center, Greenbelt, MD, United States, [4]National Aeronautics and Space Administration Postdoctoral Program, Oak Ridge Associated Universities, Oak Ridge, TN, United States, [5]Integrated Coastal Programs, East Carolina University, Wanchese, NC, United States, [6]Institute for Sustainable Communities (ISC), Montpelier, VT, United States, [7]Jet Propulsion Laboratory, California Institute of Technology, Pasadena, CA, United States

Southeast Asia is home to some of the planet's most carbon-dense and biodiverse mangrove ecosystems. There is still much uncertainty with regards to the timing and magnitude of changes in mangrove cover over the past 50 years. While there are several regional to global maps of mangrove extent in Southeast Asia over the past two decades, data prior to the mid-1990s is limited due to the scarcity of Earth Observation (EO) data of sufficient quality and the historical limitations to publicly available EO. Due to this literature gap and research demand in Southeast Asia, we conducted a classification of mangrove extent using Landsat 1-2 MSS Tier 2 data from 1972 to 1977 for three Southeast Asian countries: Myanmar, Thailand, and Cambodia. Mangrove extent land cover maps were generated using a Random Forest machine learning algorithm that effectively mapped a total of 15,420.51 km^2. Accuracy assessments indicated that the classification for the mangrove and non-mangrove class had a producer's accuracy of 80% and 98% user's accuracy of 90% and 96%, and an overall accuracy of 95%. We found a decline of 6,830 km^2 between the 1970s and 2020, showing that 44% of the mangrove area in these countries has been lost in the past 48 years. Most of this loss occurred between the 1970s and 1996; rates of deforestation declined dramatically after 1996. This study also elaborated on the nature of mangrove change within the context of the social and political ecology of each case study country. We urge the remote sensing community to empathetically consider the local need of those who depend on mangrove resources when discussing mangrove loss drivers.

KEYWORDS

mangroves, coastal ecosystem, land cover land use change, Landsat, machine learning

1 Introduction

Mangroves contribute up to 10–15% of global carbon storage for coastal oceans and up to 10–11% of biogeophysical coastal carbon cycling (Bouillon et al., 2008; Alongi, 2014; Simard et al., 2019). This makes them one of the most carbon-rich and carbon-sequestering forests with the potential to mitigate climate change and biodiversity loss (Donato et al., 2011; Howard et al., 2017; Adame et al., 2021). They are also essential to biogeochemical processes, erosion prevention, sedimentation, protection against extreme weather events, and support for coastal cultures and economies (Singh et al., 2005; Brander et al., 2012). Altogether, the ecosystem services provided by mangroves have been estimated at $1.6 billion per year (Polidoro et al., 2010). Although land managers and coastal community members have understood their value, some studies have estimated a total mangrove carbon stock decline of 158.4 Mt over the course of 1996 to 2016 (Richards et al., 2020).

Fortunately, the continuity of satellite data has enabled important insight on mangrove change dynamics. The Landsat program provides the most continuous terrestrial remote sensing records that span the last 50 years (Loveland and Dwyer, 2012; Wang et al., 2019; Yan and Roy, 2021). The Landsat repository has proven fundamental to mapping the distribution and change of mangrove forests around the world (Spalding et al., 2010; Giri et al., 2011; Hamilton and Casey, 2016; Goldberg et al., 2020; Bunting et al., 2022; Murray et al., 2022). Globally, the Landsat archive has recorded an estimated global loss of 20–35% since 1980 (Valiela et al., 2001; Polidoro et al., 2010) with estimated rates of loss between −0.16% and −3.4% (Hamilton and Casey, 2016; Bunting et al., 2022), yet many studies also document high rates of variability. One study found that various changes were often at odds with one another: in some cases even the direction of change was inconsistent among datasets Friess and Webb, 2014. Estimates of mangrove loss depend on the availability, observation period, and spatial coverage of mangrove data products (Gibbs et al., 2007; Friess and Webb, 2014) to reduce this variability. As a result, we speculate that estimates of historical rates of loss before the turn of the 21st century are not well-constrained (Everitt and Judd, 1989; Wang et al., 2019; Lewis and MacDonald, 1972; Lorenzo et al., 1979) due to four primary reasons: the challenges of working with earlier EO data (Faundeen et al., 2004; Pasquarella et al., 2016), region-specific conflicts that reduced the historical capacity for research (Han et al., 2020; Lekfuangfu and Nakavachara, 2021), the subsequent lack of remotely sensed mangrove extent data products (Wulder et al., 2016; Hu et al., 2018), and a resulting dependence on unreliable reporting of spatial extents (Friess and Webb, 2014; Wang et al., 2019). As we embark on the UN Decade on Ecosystem Restoration, we hope to enhance ongoing conversations on the historical changes of mangrove extent by filling this research gap in the literature.

The challenge with integrating earlier sensors is related to a historical lack of accessibility and limitations with the Landsat 1-5 Multispectral Scanner System (LMSS) (U.S. Geological Survey, Department 2018) and other remotely sensed observations. Individual use of Landsat imagery was severely limited until Landsat transitioned to a free and open data policy in 2008 (Pasquarella et al., 2016; Zhu et al., 2019). The high demand for Landsat data has led to improvements in calibration and corrections across various sensors, but only 49% of this satellite repository has been corrected to its highest level of precision and terrain processing (L1TP) which is characterized by its well-adjusted radiometry and inter-sensor capacity for calibration (U.S. Geological Survey, Department 2018; Yan and Roy, 2021). The remaining images in the archive have been processed to the lower L1G level of correction given the lack of elevation data and ground control references that the Level 1 Product Generation System requires (Devaraj and Shah, 2014; U.S. Geological Survey Department, 2018). Radiometric calibration that meets research standards is critical to developing the modeling methodologies that can be applied consistently over different scenes and image dates when conducting mangrove mapping projects.

As a result, most global and regional mangrove mapping efforts only date back to 2000, with a few extending into the 1990s (Bunting et al., 2022; Murillo-Sandoval et al., 2022; Hamilton and Stankwitz, 2012), which can influence conservation decision making and outcomes (Friess and Webb, 2014; Hamilton et al., 2018; Dahdouh-Guebas and Cannicci, 2021). One such study (Dahdouh-Guebas and Cannicci, 2021) made the distinction that a variety of rehabilitation and restoration targets on mangrove health assessments rely on the earliest available earth observation or past vegetation dataset to establish which areas can be sustainably rejuvenated and which are a restoration priority (Wang et al., 2019; Dahdouh-Guebas and Cannicci, 2021). Although mangrove remote sensing can be traced back to the 1970s (Kuenzer et al., 2011), they are few, the majority were completed without accuracy assessments, or were restricted to sub-regional spatial coverages (Lorenzo et al., 1979; Lewis and MacDonald, 1972; Everitt and Judd, 1989; Islam et al., 2019; Wang et al., 2019). Albeit one of the more extensive mapping efforts executed by Giri et al. (2008) were able to map country-wide mangrove changes for the tsunami affected regions of Asia (including Thailand and Myanmar) at three epochs over the course of 1975 to 2005. Furthermore, by 2018, over 435 publications had been completed enumerating the extent of mangrove ecosystems, but literature gaps remained from 1972 to 1995 (Kuenzer et al., 2011; Wang et al., 2019) with little to none of the publications utilizing wall to wall LMSS coverage for regions in Southeast Asia (Lorenzo et al., 1979; Reddy et al., 2007; Rahman, 2012; Li et al., 2013; Son et al., 2014; Islam et al., 2019).

As a consequence, our understanding of mangrove rates of change before the 1990s are variable. According to multiple comprehensive reviews on the remote sensing of mangrove extent and change (Hu et al., 2018; Friess et al., 2019; Wang et al., 2019), there is high uncertainty in both regional and global estimates. Historical and predicted estimates of change over time can therefore result in conflicting deforestation and afforestation trends depending on the datasets used in the models (Friess and Webb, 2014). Detailed reporting on mangrove area approximations are error-prone due to their dependence on coarse resolution surveys and inconsistent methods (Hu et al., 2018; Friess et al., 2019). As such, there is a need to utilize the full capacity of EO to more accurately observe mangrove forests earlier in the satellite record, particularly those with historically high and uncertain rates of deforestation.

Southeast Asia contains the greatest proportion of mangrove area (34%) in the world (Thomas et al., 2017; Bunting et al., 2022), but aquaculture, mining, agriculture, and urban expansion threaten these mangroves (Worthington and Spalding, 2018; Richards and Friess, 2016; DeFries et al., 2010; Webb et al., 2014; Friess et al., 2016). To the best of our knowledge, there are little to no studies using LMSS L1T Tier 2 Collection 1 Level 1 Raw DN observations to both map and report on the extent of mangroves for Thailand, Myanmar, and Cambodia in the 1970s. This means that national and international reporting on the net losses of mangrove extent rely on estimates from the Food and Agriculture Organization (2007; Friess et al., 2019) which contribute to contradictory deforestation rates in the mainland of Southeast Asia. For Thailand and Myanmar, studies report rates of change with a range of 7.08 ± 42.99 $km^2\ yr^{-1}$ and −60.61 ± 49.74 $km^2 yr^{-1}$ over the course of 1960 to 2010 and 1972 to 2010 (Friess and Webb, 2014). This is just one example of how high levels of uncertainty can be attributed to the use of small amounts of mangrove loss projections (Friess et al., 2019) causing them to be skewed (Ruiz-Luna et al., 2008; Kovacs et al., 2010; Friess and Webb, 2014). These case study countries were therefore selected because South Asia, Southeast Asia, and Asia-Pacific contain approximately 46% of the world's mangrove ecosystems, yet is a global hotspot of change (Gandhi and Jones, 2019). Furthermore, the study by Gandhi and Jones (2019) found that Myanmar was the primary hotspot with losses in excess of 35% from 1975 to 2005 and 28% over the course of 2000 to 2014. This study therefore chose to assess historical rates of mangrove change in the mainland of Southeast Asia where LMSS scenes were available and of sufficient quality. An older and effective mangrove extent baseline would supplement management activities with updated baselines when reporting on mangrove change dynamics (Kodikara et al., 2017; Chakraborty et al., 2019; Dahdouh-Guebas and Cannicci, 2021). Although the work of updating mangrove extent baselines will need additional studies to not run the risk of skewing future mangrove change projections

In this study, we systematically evaluate regional losses related to mangrove extent in three Southeast Asian countries to address the lack of earlier mangrove extent baselines: Myanmar, Thailand, and Cambodia. We generated a map of mangrove extent utilizing LMSS scenes that met our research criteria and was compared to existing mangrove extent data from 1996, 2007, 2010, 2016, and 2020 (Bunting et al., 2022). The implications of the new mangrove extent baseline for the 1970s is further discussed within the context of the study area's political, ecological, and economic progress and demonstrate the nuances of change specifically in these case study countries. We hope that this study can work to better inform conversations on mangrove extent and change at longer time scales and reduce the lack of data products before the 1990s.

2 Data and methods

2.1 Study area

Our region of interest (ROI) consists of three countries: Myanmar, Thailand, and Cambodia. Vietnam was originally included in the workflow, but due to the low quality and quantity of observations, the results were excluded in the study. The ROI resides within a tropical monsoon and rainforest climatic zone with temperatures above 25°C throughout the year (Peel et al., 2007). Mangrove forests are composed of trees and shrubs that are adapted to the saline and brackish conditions of the ROI coastline. They are taxonomically diverse plant species and occupy 42% of each coastline in the ROI (Bunting et al., 2022). Mangroves in Thailand are found consistently along muddy tidal flats or at the base of river mouths along the southern and eastern coasts, with a two-story forest structure (Pumijumnong, 2014), and at higher densities around the Gulf of Thailand and Andaman Sea. The lower story of mangroves in Thailand grow more than 20 m and are dominated by species from the *Rhizophora, Herritiera*, and *Xylocarpus* genera. The upper story mangrove species of Thailand include the *Bruguiera* and the *Ceriops* genera, with some of these species like the *Bruguiera gymnorrhiza* growing more than 40 m above the forest and with trunks as thick as 2 m (Aksornkoae, 2012; Pumijumnong, 2014). Myanmar hosts an array of mangrove species just as numerous and diverse from the *Rhizophora*, *Avicennia*, *Bruguiera*, *Ceriops*, and *Xylocarpus* (Zöckler and Aung, 2019) genera and are primarily found in the Rakhine, Ayeyarwady, and Tanintharyi divisions (Zöckler and Aung, 2019). Mangrove forests in Cambodia are found primarily in four provinces, Koh Kong, Sihanoukville, Kampot, and Kep (Nop et al., 2017; Kozhikkodan Veettil and Quang, 2019). The most found species in Cambodia are from the *Rhizophora*, *Nypa*, *Bruguiera*, *Ceriops*, *Lumnitzera*, *Heritiera*, *Xylocarpus*, *Hibiscus*, *Phoenix*, and *Acrostichum* genera.

2.2 Pre-processing and classification

The methodology (Figure 1) used to produce the historical extent maps was divided into six steps – delineation of the ROI, evaluation and selection of Landsat 1-2 MSS (LMSS) Collection 1 Tier 2 DN, processing and correction of LMSS, generation of a 1970s mangrove map, assessment of accuracy, and a comparison of the baseline to Global Mangrove Watch (GMW) mangrove extents (Bunting et al., 2022). All processing and analyses were carried out using the Google Earth Engine Platform, ArcGIS Pro, and R. Figure 2 displays the coastal mask used to delineate the region of interest, pixels assigned with a value of zero due to no Landsat observations, the Landsat scene WR-1 path and row, and the 1970s mangrove extent. The final LMSS composite and the 1970s mangrove classification can also be referred to in Figures 1, 3.

Step one in our workflow included data filtering and selection followed by a series of pre-processing steps. To delineate coastal areas and subset the LMSS data needed for processing, a coastal mask was generated to include all potential mangrove areas. The

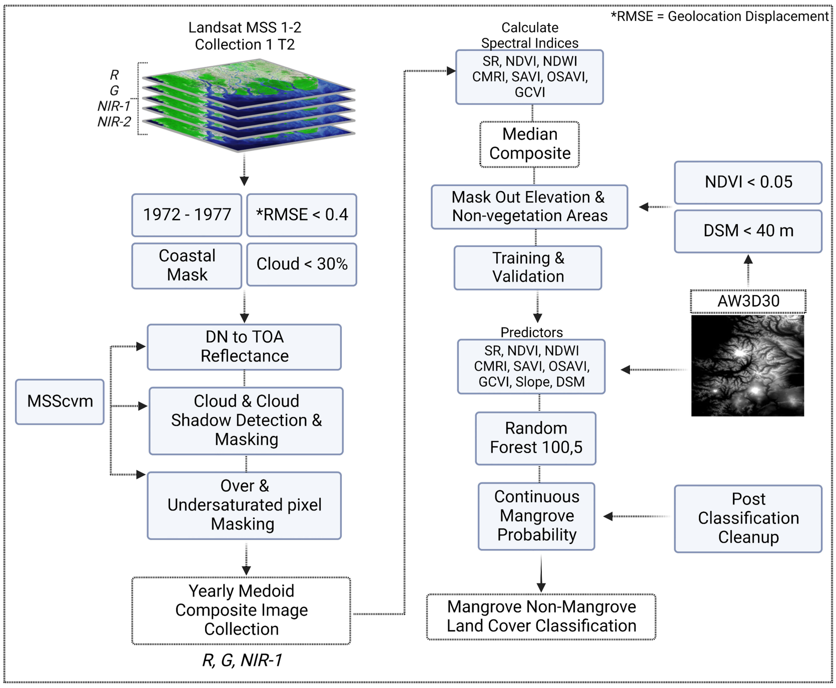


FIGURE 1
Overview of the methods used to conduct a random forest classification of mangrove forest extent in the case study countries. The analysis consisted of filtering and pre-processing; creation of yearly image composites; calculation of predictors; masking to constrain the analysis; a random forest classification; and post-classification cleanup.

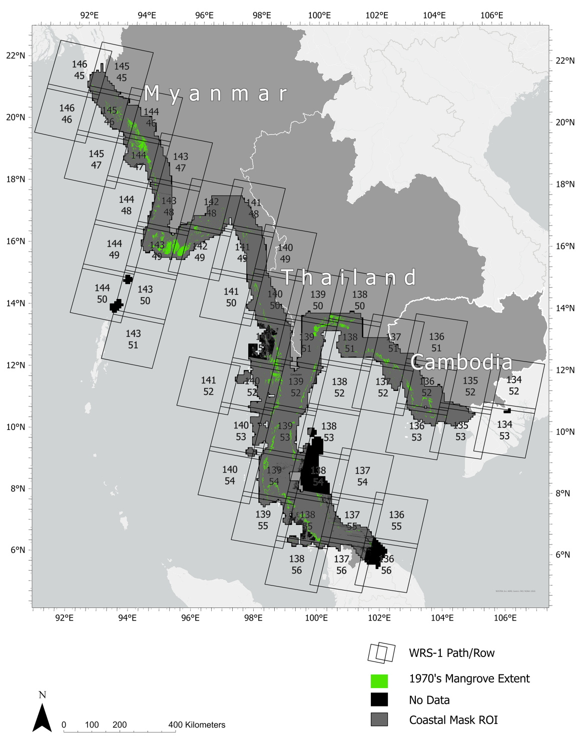


FIGURE 2
The study area showing the coastal mask that was generated to constrain the analysis to mangrove areas, the relevant WRS-1 Path/Row of the Landsat images used, the areas with no available Landsat data, and the 1970s mangrove distribution.

mask included: the known extent of more current mangroves and coastal wetlands based on the global Wetland Extent Map (Mcowen et al., 2017) and the GMW dataset (Bunting et al., 2022), areas within 10 km of the shoreline, based on the global shorelines data (Sayre et al., 2019) and areas lower than 20 m elevations based on the Shuttle Radar Topography Mission (SRTM) DEM (Farr et al., 2007; Yang et al., 2011).

Given the unavailability of Tier 1 Landsat data, we utilized LMSS Tier 2 data for our analysis. Scenes with this level of processing have a lower radiometric and positional quality, but due to the lack of available scenes, this collection was selected. The coastal mask was used to select scenes from LMSS Tier 2, which were scaled and calibrated to at-sensor radiance. Only systematic terrain (L1GT) and systematic (L1GS) processing were applied to this collection due to insufficient ground control, excessive cloud cover, and geolocation issues (U.S. Geological Survey Department, 2018). This collection has a resampled spatial resolution of 60 m and a spectral range of 0.5–1.1μ including the Green, Red, NIR-1, and NIR-2 channels, and corresponds to WRS-1 (Wulder et al., 2022; U.S. Geological Survey Department, 2018). All scenes available from 1972 to 1977 were selected using the coastal mask as a spatial filter, which was then followed by a series of exclusionary steps. Scenes that did not have all of the five bands present which are the Red (B4), Green (B5), Near Infrared 1 (B6), Near Infrared 2 (B7), and the quality assurance bitmask (QA_Pixel) bands were excluded. Additionally, scenes were excluded if they had L1GS processing, exceeded a spatial displacement greater than ~24m or a root mean square value of 0.4, and cloud cover greater than 30%; effectively reducing imagery from 3,153 to 689 (see Figure 4).

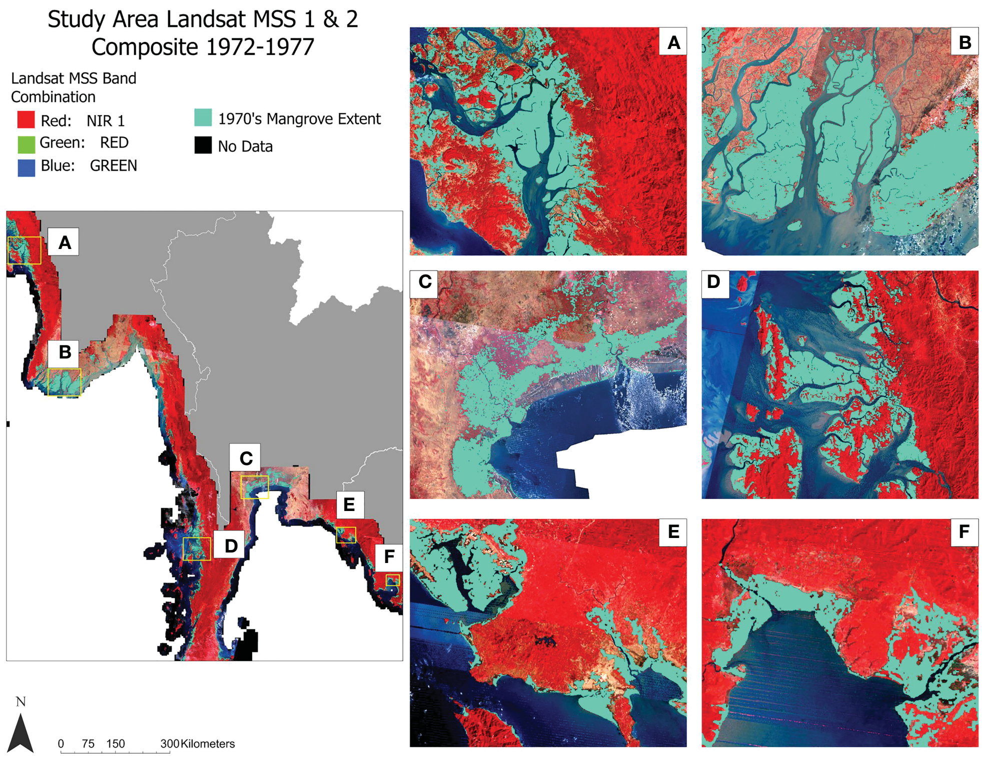


FIGURE 3
The LMSS composite circa 1972 to 1977. The displayed LMSS mosaic was constructed using the medoid on a yearly basis and then the yearly mosaics were composited using the median across the temporal period. The resulting random forest classification of mangrove extent is overlaid on the composite. Different mangroves across the study area are highlighted which include the **(A)** Rakhine state, Myanmar, **(B)** Ayeyarwady Delta, Myanmar, **(C)** Vicinity of Samut Sakorn, Thailand, **(D)** Tanintharyi Division, Myanmar, **(E)** Trat, Myanmar bordering Cambodia, and **(F)** Koh Kong Province, Cambodia.

Images were then manually excluded if they had excessive cloud cover over mangrove areas, erroneously saturated pixels, or abnormal image artifacts, further reducing the collection to 371 images (see Figure 5). These issues were related to excessive detector striping, transcription artifacts, abnormal saturation, memory effect, and scan mirror pulse errors (U.S. Geological Survey Department, 2018; Vogeler et al., 2018).

Following the manual inspection and removal of Landsat scenes from Landsat MSS over the study area, the next step was to mask out cloud cover and correct the imagery to top-of-atmosphere reflectance. We did this by using an automated cloud and cloud shadow detection and masking algorithm proposed by Braaten et al. (2015), called the Landsat Multispectral Scanner System clear-view-mask or MSScvm. This algorithm is an already established automated approach that identifies and masks out clouds based on green band brightness and the normalized difference between the green and red bands. It further

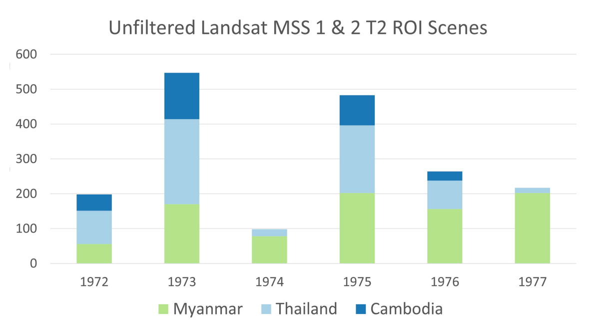


FIGURE 4
A histogram showing the image counts for all available Landsat 1 & 2 imagery over the region of interest and over the years of 1972 to 1977. The histogram shows that image availability was high before the filter process was initiated.

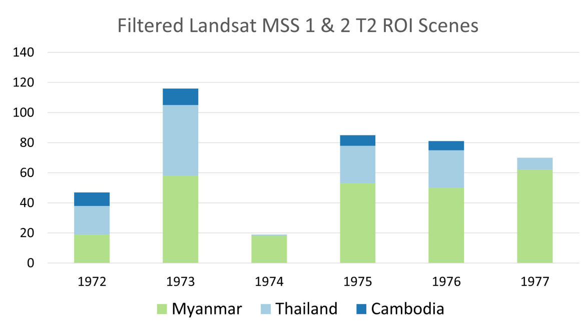


FIGURE 5
A histogram showing the image counts for all available Landsat 1 & 2 imagery over the region of interest and over the years of 1972 to 1977. Image availability was decreased by 88.23% once filters were applied to the collection to exclude imagery based on RMSE, cloud cover %, spatial characteristics, sensor attributes, and scene processing.

identifies and masks cloud shadows using near infrared band darkness and cloud projection. Along with this cloud detection, it also corrects for topography-induced illumination and water identification using a digital elevation model (Braaten et al., 2015). Then on a per image basis, MSScvm was applied to each image effectively removing the majority of cloud cover and cloud shadow. Following that, we created a mosaic for subsequent classification. As shown in Figures 4, 5, the amount of imagery is highly variable. This study selected the years 1972 to 1977 due to these years having the highest and most semi-consistent image count with almost wall to wall coverage (see coastal mask in the Figure 1 study area) during the 1970s. Once the images were identified, the cloud free mosaics were generated per calendar year from 1972 to 1977 on a per pixel basis using the medoid, a more robust version of the median. Although these studies are more effective with seasonal composites, this was not feasible due to low image counts. The medoid represents the point with the minimal summed distance to all points in a data set. It also takes into consideration the multiple dimensions in the selection relevant to the different bands of a scene found within a year (Flood, 2013). The medoid was calculated by finding the difference between the median and the corresponding median of each band squared followed by finding the sum of the powered differences across bands per image for all observations. These annual medoid mosaics were much more sensitive to extreme outliers resulting in a more consistent representation of the study area. This approach was also selected to account for the inconsistent number of Landsat observations per year. The yearly medoid image collection was then used to produce a single five-year composite over the study period using the median across all medoid observations (see Figure 3 for composite results and Figure 1 for workflow).

Following the steps used to create a single mosaic of LMSS from 1972–1977, a series of indices were calculated that are ideal for mapping mangrove extents (Yancho et al., 2020). The indices that were calculated include the simple ratio (SR), normalized difference vegetation index (NDVI), the normalized difference water index (NDWI), the combined mangrove recognition index (CMRI), the soil-adjusted vegetation index (SAVI), the optimized soil-adjusted vegetation index (OSAVI), and the greenness chlorophyll vegetation index (GCVI) which serves as a proxy for chlorophyll content and proved to be useful in some mangrove mapping studies (Jordan, 1969; Tarpley et al., 1984; Huete, 1988; Gao, 1996; Gupta et al., 2018; Yancho et al., 2020; Rondeaux et al., 1996; Gitelson et al., 2003; Chamberlain et al., 2021). In addition to the vegetation indices, slope and elevation were also incorporated which was extracted from JAXA's Land Observing Satellite (ALOS) (Takaku et al., 2014; Takaku et al., 2020). The predictors used for the random forest classifier can be referred to in Table 1 in addition to the original bands used in the LMSS collection.

Before beginning the process of collecting model calibration and validation samples, masks were applied to the coastal region of interest (see Figure 1) using the ALOS DSM (Takaku et al., 2014; Takaku et al., 2020) and NDVI bands. These datasets were used to exclude areas that were not vegetation using NDVI pixels less than 0.05 and areas of elevation greater than 40 m. This masking excluded any area that did not have an NDVI values associated to dense vegetation, such as urban, water, and bare ground areas in addition to higher elevation areas. Given this level of masking, this study specifically focused on dense assemblages of mangrove forests and excluded only the most fragmented mangroves with low NDVI values. Training polygons were collected using the final composite by digitizing areas representing the most homogenous mangrove and non-mangrove vegetation pixels. Due to the lack of reference data from the 1970s, the composite itself was used as a reference to digitize the training samples. In total, 1,134 points were selected for the mangrove (n=283) and non-mangrove (n=851) land cover classes. Using the Landsat MSS composite (see Figure 3) an area was designated as mangrove if it was found within the coastal mask

TABLE 1 The vegetation index inputs were created using the bands from Landsat MSS 1 & 2 as listed on the first row.

Variable	Spatial & Temporal Resolution	Dataset	Reference
Green, Red, NIR-1, NIR-2, Pixel_QA	60 m, 1972–1977	Landsat MSS 1 & 2	(U.S. Geological Survey Department, 2018; Gorelick et al., 2017)
*NDVI	60 m, 1972–1977	Landsat MSS 1 & 2	(Tarpley et al., 1984; U.S. Geological Survey Department, 2018)
*NDWI	60 m, 1972–1977	Landsat MSS 1 & 2	(Gao, 1996; U.S. Geological Survey Department, 2018)
*GCVI	60 m, 1972–1977	Landsat MSS 1 & 2	(Gitelson et al., 2003; U.S. Geological Survey Department, 2018)
*SR	60 m, 1972–1977	Landsat MSS 1 & 2	(Jordan, 1969; U.S. Geological Survey Department, 2018)
*CMRI	60 m, 1972–1977	Landsat MSS 1 & 2	(Gupta et al., 2018; U.S. Geological Survey Department, 2018)
*SAVI	60 m, 1972–1977	Landsat MSS 1 & 2	(Huete, 1988; U.S. Geological Survey Department, 2018)
*OSAVI	60 m, 1972–1977	Landsat MSS 1 & 2	(Rondeaux et al., 1996; U.S. Geological Survey Department, 2018)
*Slope	2006	Advanced Land Observing Satellite	(Takaku et al., 2014; Gorelick et al., 2017; Takaku et al., 2020)
*DSM	30 m, 2006	Advanced Land Observing Satellite	(Takaku et al., 2014; Gorelick et al., 2017; Takaku et al., 2020)

The slope was calculated using the DSM from the ALOS dataset. All the datasets were accessed using the Google Earth Engine repository. The slope was calculated using the Google Earth Engine slope functions (Gorelick et al., 2017). The predictors used for the Random Forest model used in this study are denoted by an asterisk.

region, was found along the coastline, found along tributaries closest to the coastline, had a dense texture, and was a patch with high and consistent NDVI values. The non-mangrove training areas were selected based on whether it was found in water, areas at greater distances from the coast, and in groupings of pixels that were not homogenous or did not have a high NDVI value. These characteristics were chosen given that the minimum mapping unit (MMU) was around 0.16 ha and that we had a goal of mapping an assortment of mangrove trees and not individual mangrove trees below the MMU.

Following the preparation of predictors and collection of training samples, a random forest algorithm was used to predict the distribution of mangrove and non-mangrove areas for the entire study area using the LMSS mosaic. Random forest is an approach that uses non-parametric classification and decision trees in addition to classification and regression trees (Breiman, 2001). The hierarchy of this classifier is composed of a root node, inclusion of predictor samples, node separator with relevant decision rules, and the end of the leaf node with the desired classes – or in our case the probability of belonging to the mangrove class. Random forest was also chosen because models in other studies resulted in higher accuracies in comparison to other decision tree classifiers (Breiman, 2001; Pal, 2003; Ghimire et al., 2012; Belgiu and Drăguţ, 2016). The predictors that were selected for the model used in this study included the previously prepared indices or elevation parameters; SR, NDVI, NDWI, CMRI, SAVI, OSAVI, GCVI, and DSM. The random forest model we selected utilized a total of 100 trees sampled at random for every 5th predictor, a minimum leaf population of 1, a bag fraction of 0.5 per tree, no limit on the maximum number of leaf nodes in each tree, a randomization seed value of 0, with the output mode set to a probability output. Following this classification of continuous mangrove probability, a series of post classification steps were used to remove noise and to threshold the classification's bimodal distribution. First the classification was automatically thresholded using a gray level histogram detection method proposed by Otsu (1979) to divide the layer into two distinct classes, mangrove, and non-mangrove. Once the classification was automatically converted into a binary classification of mangrove and non-mangrove areas, the classification was cleaned up to remove noise using a majority filter. The majority filter was applied using a 3 by 3 kernel majority filter where a given pixel would be changed if most of the cells within a neighborhood were contiguous. Following the majority filter, a series of dilation and erosion techniques were used to generalize the zones using an evaluation of immediate orthogonal and diagonal neighbors for a given mangrove or non-mangrove region (Serra, 1982). The order of sorting priority was based on the size of the mangrove and non-mangrove zones when performing the classification smoothing. The classification was manually cleaned up to eliminate additional salt and pepper areas, areas that had visible errors introduced by excessive cloud cover or had any additional noise. An example of this would be any errors of commission found farther from the coastline where brackish waters are less concentrated.

To assess the classification, we followed the "good practices" proposed by Olofsson et al. (2013) and Olofsson et al. (2014) using a random stratification approach over the study area. It is important to note that the area proportion for the mangrove and non-mangrove class were 16% and 84% of the total study area and the sample allocation was based on the smaller area proportion class (mangrove). Furthermore, we anticipated an accuracy of 70% and a proportion of 20% for the mangrove class, and a target standard error of 1%. This leveraged a set of 300 samples for the non-mangrove class and a set of 67 samples for the mangrove class. However, a total of 28 were impossible to verify visually, therefore our final dataset resulted in 339 samples: 54 to mangrove and 285 to non-mangrove. The verification process took place in the Collect Earth platform (Bey et al., 2016; Saah et al., 2019), each sample was inspected using both the original LMSS mosaic, Google Earth Imagery (Lisle, 2006), and the classification overlaid as a reference. Sample points were validated as mangrove when 50% of a 30 m x 30 m square centered about a sample point had a higher NDVI reflectance (manifested as a dark red color), were adjacent to the coastline, at the intersection of a river outlet and the ocean, or were greater than the MMU of ~0.16 ha. The non-mangrove class was labeled if a given sample point was found in open water, bare ground with some vegetation cover, water with some vegetation, heavily fragmented vegetation, and vegetation that was not immediately adjacent to the coastline or river outlets for greater than 50% of a 30 m x 30 m area. Then the attribute table of the validation point was updated by extracting the value found in the classification at each point location. This was used to calculate the error matrices, overall, producer's, user's accuracy, and Kappa Coefficient.

Once the classification and accuracy assessment was completed, we analyzed the extent of mangrove ecosystems for our ROI within the context of each country's unique circumstances, GMW extents, and other external estimates. In order to measure change in mangrove extent between the 1970s and 2020, we compared our 1970s mangrove cover map to maps created by GMW for 1996, 2007, 2010, 2016, and 2020 (Bunting et al., 2022). The GMW maps were reprojected and resampled to match the spatial resolution of the LMSS classification results. Then, the layer was rasterized. Lastly, the GMW raster was masked to exclude all areas that overlapped with no Landsat observations available in the final LMSS classification product.

3 Results

3.1 Data processing and uncertainty

A total of 3,153 images were identified during the initial data exploration phase before additional filters were applied. This study identified 371 images suitable for classification that met spatial offset and cloud cover filters, which reduced the available imagery by 88%. The study had an average of 63 ± 38 images per year but with significant interannual variability. The highest image availability occurred in 1973 (n = 109) and 1976 (n = 103). The year with the lowest image count was 1974 with 9 images. Following the filtering and scene selection phase, the LMSS imagery was corrected to top-of-atmosphere, cloud and cloud shadow masked, and then composited into a mosaic over the study period resulting in a cloud free medoid mosaic of LMSS. This approach likely has high amounts of variability and uncertainty due to the lack of observations & the inability to map mangrove cover seasonally. Additionally, the coarser spatial, spectral, and radiometric resolution of MSS constrains the capacity for mapping mangroves during the 1970s. This work would greatly benefit from additional efforts to map mangroves on a seasonal basis over a longer time period consistently with harmonization across all of Landsat's sensors as done by other studies (Braaten et al., 2015; Zhu et al., 2018; H. Nguyen et al., 2019; Yan and Roy, 2021). Following the classification, the final mangrove data product was assessed for accuracy, but with other constraints.

The overall accuracy and kappa coefficient was 95% and 0.82 respectively. Producer's and User's accuracy were 98% and 96% for the non-mangrove class and 80% and 90% for the mangrove class (Table 2). These accuracy results indicated that the model was less likely to identify real mangrove features on the ground in comparison to the mapped mangrove feature in the map. The higher reliability, or User's accuracy for the mangrove class indicated that map users were more likely to find the mangrove areas identified in the map on the ground. Some of these uncertainties were likely related to the quality of the mosaic and how consistent a given pixel may be. Take for example the issue of low image counts for 1974 in this study and the fact that there were regions with no Landsat observations.

TABLE 2 Accuracy metrics.

	Producer Accuracy	User Accuracy
Non-mangrove	0.982	0.962
Mangrove	0.796	0.895
Overall Accuracy	0.953	
Kappa statistic	0.815	

3.2 1970s baseline of mangrove extent and assessment of change from 1970s to 2020

Our 1970s map of mangrove extent identified 15,420 km^2 across our study area (Table 3). Myanmar had the greatest extent of mangroves (9,272 km^2), followed by Thailand (5,407 km^2) and Cambodia (742 km^2). When comparing our new 1970s baseline data to existing GMW maps of mangrove extent, we found a sharp decline in mangrove area between the 1970s and 1996. Across the study region, mangrove area declined by 47% (corresponding to a loss of 8,239 km^2) during this period (Table 3). Loss rates were highest in Thailand (58%) and lowest in Cambodia (14%). In Thailand, declines in mangrove areas were especially pronounced around Bangkok and along the eastern portion of the Gulf of Thailand (Figure 6). In contrast, areas with the greatest occurrence of persistence and gain were found in the Mu Ko Phayam National Park along the coastline up to the Mu Ko Ra-Ko Phra Thong National Park. Myanmar experienced a 42% decline in mangrove area between the 1970s and 1996. Here, loss was greatest around the Ayeyarwady Delta (Figure 7). In Cambodia, most losses were concentrated along the Koh Kong coastline in the northwestern portion of the country (Figure 8). Although Cambodia experienced overall declines in mangrove extent between 1970s and 1996, there were some southwest gains in the Bay of Kampong Som in the Botum Sakor District of Koh Kong Province (Figure 8).

Firstly, the GMW estimates were masked using the areas labeled as 0 due to a lack of Landsat observations in the LMSS mosaic. The inventory of extent of different time points up to 2020 show that rates of mangrove loss across our study region declined dramatically after 1996 (Table 3; Figure 9). Overall, mangrove extent declined by 7% between 1996 and 2007 and these rates were similar across the three countries. Mangrove extent then showed little change between 2007 and 2016. Between 2016 and 2020, mangrove extent increased in all three countries; rates of mangrove expansion ranged from 7% (Cambodia) to 18% (Thailand). These results further extend our understanding of mangrove change before the 1990s, a period of data scarcity when it comes to mangrove data products.

4 Discussion

4.1 Data processing challenges and limitations

In this study, we provided a semi-automatic approach to delineating the extent of mangrove area using LMSS data from the 1970s. We leverage the earliest available Landsat imagery using

TABLE 3 A comparison of extent estimates from the new 1970s baseline and the *GMW extent.

	MMR km^2	Net Loss/ Gain km^2	THAI km^2	Net Loss/ Gain km^2	KHM km2	Net Loss/ Gain km^2	Total km^2	Net Loss/ Gain km^2
1972	9271.98	–	5406.59	–	741.95	–	15420.51	–
*1996	5345.00	−3926.98	2259.00	−3147.59	636.00	−105.95	8240.00	−7180.51
*2007	4965.49	−379.51	2119.24	−139.76	596.37	−39.63	7681.11	−559.00
*2010	4942.92	−22.57	2142.58	+23.34	589.83	−6.54	7675.33	−5.78
*2016	4912.69	−30.23	2144.17	+1.59	587.61	−2.22	7644.47	−30.86
*2020	5435.39	+523	2527.99	+383.82	626.92	+39	8590.30	+946

GMW was masked to exclude areas that overlapped with no Landsat observation areas in the LMSS mosaic.
* Each estimate is attributed to global mangrove watch extent estimates (Bunting et al., 2022).

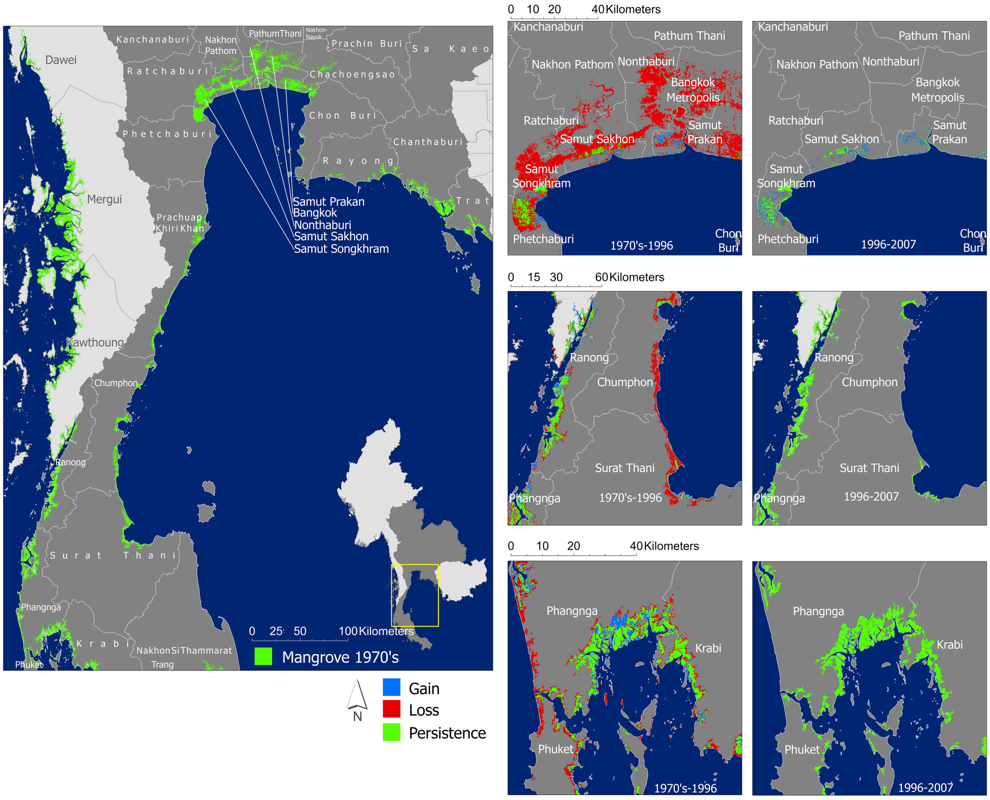


FIGURE 6
The classification difference between the 1970s Thailand mangrove baseline and the 1996 GMW mangrove extent. The losses were most significant in the Samut Sakhon, Nonthaburi, Bangkok, and Samut Prakan.

MSS and compared it to more recent mangrove extent maps to understand historical mangrove change. However, some limitations remain, for instance none of the LMSS data from this period has been processed to the highest level of terrain and precision (Braaten et al., 2015; Roy et al., 2016). The issues that affected the quality of the collection, a sensor that makes up a significant portion of the early Landsat record (Yan and Roy, 2021), were related to a variety of anomalous errors such as memory, effect, scan correlated shifts, and scan mirror pulse[1]. In addition to sensor-related issues, there is also the difficulty of conducting a remote sensing investigation in the case study countries, one of the planet's cloudiest regions (Kontgis et al., 2015; Li et al., 2018; Oliphant et al., 2019). Because of these challenges, this study excluded imagery that had a spatial offset greater than 24 m, cloud cover of less than 30%, imagery with oversaturated pixels, or excessive striping. It should also be noted that the number of images found per year was not consistently the same (Figures 4, 5) or of an ideal quantity to do annual image classifications and change detection. As a result, we had to composite multiple years' worth of data to cover the entire study area. These challenges will make it difficult to expand this approach to other countries outside of our study area.

Following this phase, careful measures were taken to identify the LMSS scenes that would be used in this study. The images then had to be preprocessed to TOA and cloud masked using the MSScvm (Braaten et al., 2015) algorithm. This algorithm was developed by researchers to overcome the challenge of automatically detecting and masking clouds from MSS (Braaten et al., 2015), but with some limitations. This algorithm was designed to work with temperate ecosystems, but due to the lack of automatic algorithms for MSS cloud detection, MSScvm was selected for pre-processing procedures. Many measures were taken to identify the best approach to establishing a new baseline, but these limitations must be considered.

4.2 Shifting perspectives on mangrove change

With our new 1970s baseline map of mangrove extent in our case study countries, we identified a sharp decline in mangrove extent primarily for Myanmar and Thailand between the 1970s and 1996. However, after 1996, mangrove loss rates declined dramatically, and mangrove extent has been relatively stable since the mid-2000s according to this assessment. We do believe that an additional effort must be done to map the extents more consistently and harmoniously across sensors over a dense time series. But that was not possible at this time because of the lack of MSS scenes of sufficient quality. The proximate and underlying drivers of gains and losses for our study area are complex due to the history of political and economic instability from situations like the reign of the Communist Party of Kampuchea (CPK) over the course of 1975

1 https://www.usgs.gov/landsat-missions/landsat-known-issues

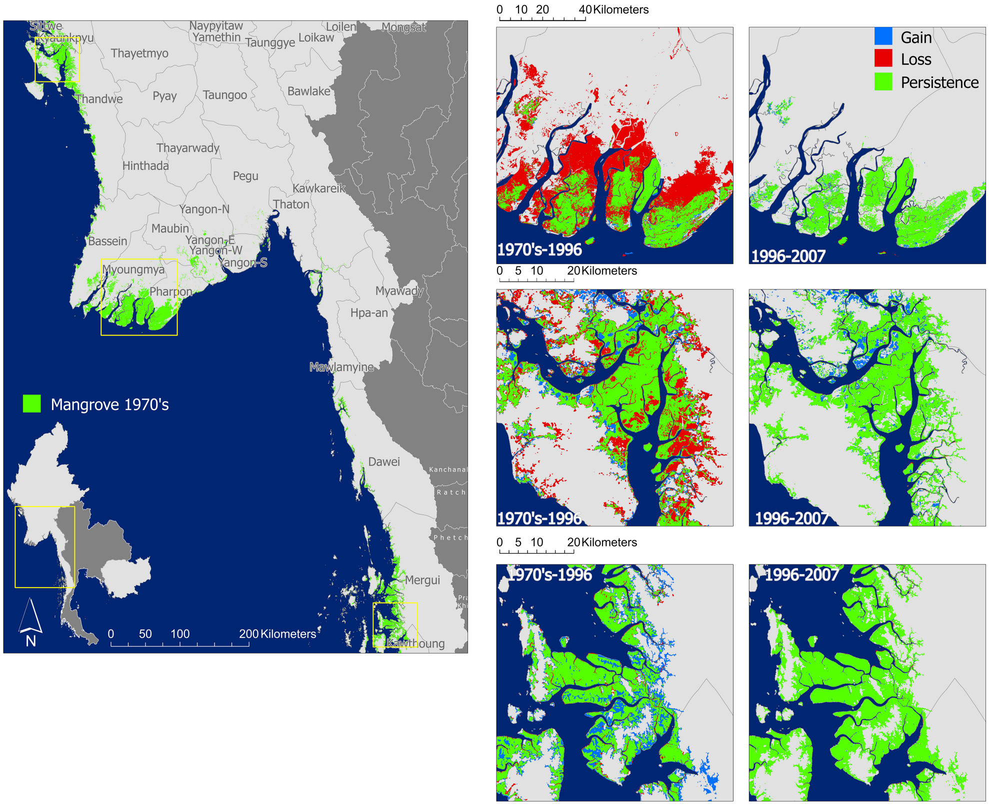


FIGURE 7
The classification difference between the 1970s Myanmar mangrove baseline and the 1996 GMW mangrove extent. The area of Myoungmya and Pharon showed the most jarring losses in Myanmar.

to 1979 in Cambodia (Mosyakov, 2000) or the lack of electricity and subsequent need for mangrove firewood (from the Ayeyarwady delta) in the Yangon province of Myanmar. Furthermore, studies have indicated that some of the Myanmar and Cambodia mangrove losses may have been attributed to Thailand's ban on logging in 1989 (Brown et al., 2001) proving that conservation measures can have unintended deforestation consequences (Brown et al., 2001; Pumijumnong, 2014; Lim et al., 2017). Meanwhile, Cambodia's decades of political conflict under the reign of CPK may have resulted in lower rates of deforestation due to the lack of timber demand prior to the 1980s (Boutros-Ghali, 1994; Le Billon, 2000). Studies on the political ecology coupled with this study can help shed light on the transition from war to peace or the nature of different industries and their impacts on land cover and land use change.

4.3 Country specific perspectives

In Cambodia, we calculated a total of 742 km^2 for the period of 1972 to 1977. This is a more conservative estimate compared to other studies that inventoried a total of 946 km^2 for the same period without using remote sensing (Cambodia Land Cover Atlas 1985/87–1992/93, 1994; Cambodia Forestry Policy Assessment, 1996; Ministry of Environment, Kingdom of Cambodia, 2009). Following this time period, one study estimated a total of 758 km^2 for 1989, while another study mapped a total of 646 km^2 by 1996 (Kozhikkodan Veettil and Quang, 2019; Bunting et al., 2022). Over the course of 1996 to 2016, different studies estimated a mangrove loss of 208–300 km^2. When comparing the new 1970s baseline to the estimate by Bunting et al. (2022) in 2020, this study indicates an additional loss of 115 km^2 ± 174 km^2. Many of the drivers of degradation and change have been attributed to shrimp pond aquaculture; salt pan and charcoal production; and infrastructure development (Kozhikkodan Veettil and Quang, 2019), but we suggest that the drivers of change were also very much political especially during the temporal period of this study (Boutros-Ghali, 1994; Le Billon, 2000).

It is important to note that several studies have indicated a high level of uncertainty on Cambodian mangrove forest estimates (Rizvi and Singer, 2016; Nop et al., 2017; Kozhikkodan Veettil and Quang, 2019). Although there is some uncertainty, these statistics show a trend of little change between the 1970s era up to 1996 and may serve as additional support to other claims on the political drivers of mangrove persistence. During the period of 1975 to 1979, the CPK (Boutros-Ghali, 1994; Le Billon, 2000) was the ruling regime in Cambodia. During this time, the country experienced severe famine, deaths associated to the lack of medicine, the proliferation of disease, and mass genocide which led to the deaths of up to 1.5 to 2 million people (Locard, 2005). These extremely difficult circumstances indirectly enabled forest stability and even gains (Le Billon, 2000) throughout the 1970s, which

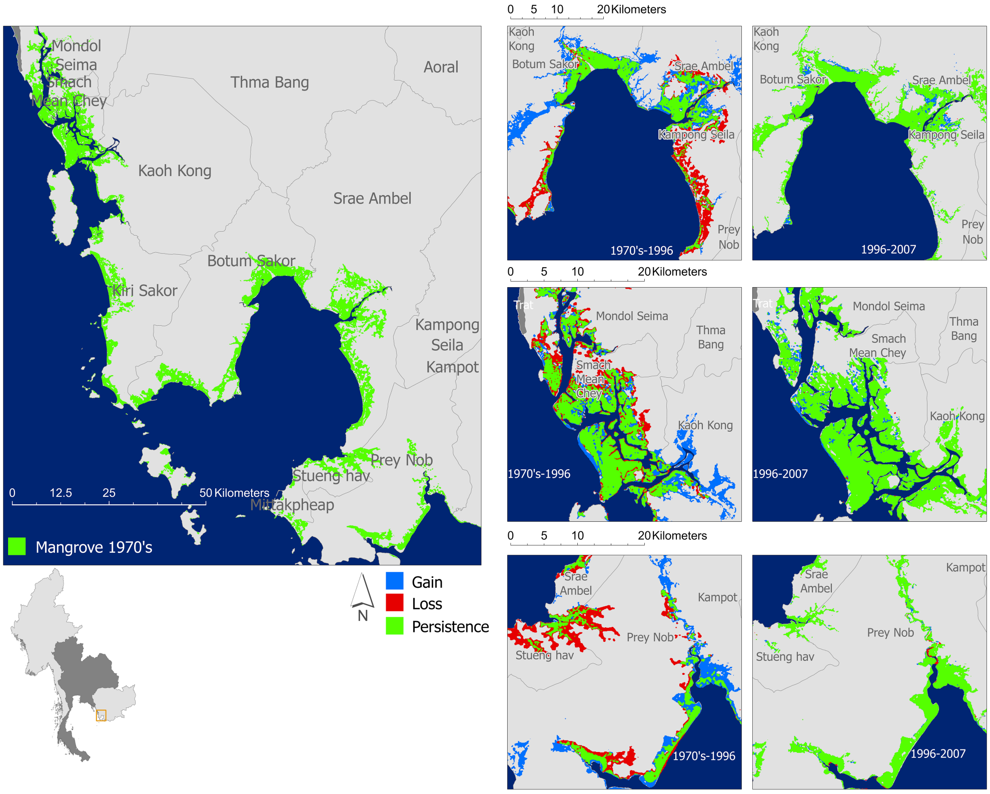


FIGURE 8
The classification difference between the 1970s Cambodia mangrove baseline and the 1996 GMW mangrove extent. The areas of Botum Sakor showed a trend of persistence, while Srae Ambel showed areas of mostly loss.

helped almost two-thirds of the country to be completely forested by the early 1980s (Cambodia Land Cover Atlas 1985/87–1992/93, 1994). Also, the aquaculture industry was not actively introduced until 1960 in Cambodia and was relatively inactive during the time of the CPK conflict. The active political conflict coupled with the lack of aquaculture activities are the likely drivers of mangrove stability seen between the 1970s to the 1990s.

Our estimate of mangrove extent from the 1970s for Thailand (5,407 km^2) were somewhat higher than estimates from previous studies that mapped an extent with a range of 3,127–3,679 km^2 (Aksornkoae, 2012; Pumijumnong, 2014; Charupphat and Charupphat, 1997; Klankamsorn and Charuppat, 1982; Barbier and Cox, 2002; Naito and Traesupap, 2014). However, the large losses in mangrove extent that we observed between the 1970s and 1996 agree with other studies (Charupphat and Charupphat, 1997; Klankamsorn and Charuppat, 1982; Aksornkoae, 2012; Naito and Traesupap, 2014). One such study that documented the economic and demographic drivers of mangrove losses over the course of 1975 to 1996 in Thailand indicated that almost half (46%) of all mangroves were lost to coastal shrimp farming (50–65%) and the increased demand for land in coastal areas due to urbanization and agriculture during the period of 1979 to 1996 (Barbier and Cox,

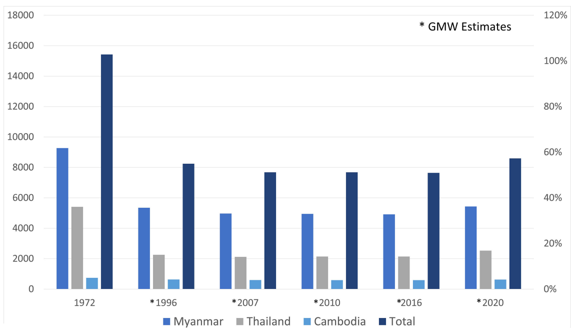


FIGURE 9
A histogram showing the distribution of mangrove extents comparing the new 1970s era baseline to the estimates for GMW at different time points; 1996, 2007, 2010, 2016, 2020. These estimates indicated an average loss of 44% from the 1970s to 2020 over the study period.

2002; Spalding et al., 2010). This study indicates that the losses were 12% greater (58%) over the same time.

Mangrove losses associated with Thailand were primarily related to the demand for land development, aquaculture, economic incentives, and policy failures, but only up until industries started to aggressively expand. The studies by Barbier and Cox (2002) and Bantoon (1994) observed how even though aquaculture was introduced as early as 1974, the industry began to have severe mangrove loss impacts after 1985 due to Japanese demand (Bantoon, 1994; Barbier and Cox, 2002). In addition, sustained economic growth caused coastal populations to grow, boosting the demand for urbanization and economic crowding around mangrove areas. This caused shrimp production to go from 15,000 metric tonnes (KMT) in 1985 to 264,000 KMT in 1994 (Pednekar, 1998; Barbier and Cox, 2002). It also caused the shrimp farm area to rapidly expand between 1983 and 1996 and the number of farms to increase from 3,779 to 21,917. This boom in the aquaculture industry in addition to increased urbanization are the likely contributors to the severe mangrove losses seen throughout Thailand.

Previous estimates of mangrove extent and change in Myanmar exhibit a large amount of variation, highlighting the need for better data for this region (Aung, 2007; Spalding et al., 2010; Thant et al., 2012; Webb et al., 2014; Giardino et al., 2016; Veettil et al., 2018; Alban et al., 2020). Some of these claims documented that extreme overexploitation began as early as the Second World War to satisfy the demands of the military with the worst forest overexploitation occurring over the period of 1949 to 1972. According to Oo (2002) and Kyi (1992), mangrove forests in the Ayeyarwady delta decreased from 2,345 km^2 in 1954 to 1,786 km^2 in 1984. This study speculated losses that are much higher than our findings which established an area of 9,272 km^2 in the 1970s. More recently, Bunting et al. (2022) mapped a total of 5,821 km^2 of mangroves across the entire country in 1996, while Alban et al. (2020) mapped a total of 13,233 km^2 for the same year. These contradictory statistics demonstrate the need for region-specific mapping approaches in lieu of sub-setting a global dataset to report on country specific mangrove extent (Estoque et al., 2018; Alban et al., 2020). Also, Alban et al. (2020) estimated higher rates of mangrove deforestation in Myanmar, with 60% of mangroves permanently or temporarily lost between 1996 and 2016 due to the cultivation of rice, oil palm, and rubber in addition to urbanization.

Other studies that were done during the 1990s indicated additional land use drivers of change were to blame for the severe losses that were also found in Myanmar in the three main mangrove regions: Ayeyarwady, Tanintharyi, and Rakhine. According to Oo (2002), the main objective of mangrove management was fuel-wood and charcoal production. Then during Myanmar's period of insurgency (1949–1972), the forest department was not able to effectively use forest management at large. However, mangrove species were not as affected by commercial demand, but by local demand. For example, the more heavily populated area of Yangon relied heavily on mangrove for charcoal and firewood production from the Ayeyarwady area, resulting in heavy losses. The Rakhine state on the other hand is an area with lower population and was speculated to have minimized losses according to Oo (2002) in the early 2000s. Higher populations and the demand for charcoal and fuel wood production therefore led to more extensive mangrove losses according to these studies. Following such losses, the Forest Department started intensive mangrove planting projects in 1975 followed by strict prohibitions of mangrove-derived charcoal and firewood after 1993. An important distinction that the study by Oo (2002) made is that many of the areas were experiencing losses due to the lack of electricity and need for energy which was an indirect driver of the fuel wood and charcoal production industry before prohibition.

Much of the losses of mangroves in our study region were driven by economic concerns. The Ayeyarwady delta of Myanmar experienced losses to mangrove charcoal and firewood production, but this resource was used to address the complete lack of electricity in the adjacent area of Yangon. This study indicated that almost half of mangrove area was lost by 1996, but we also urge researchers to have a sensitive perspective on the drivers behind the loss. Yes, these losses were extensive, but these results should be considered within the context of the people of Myanmar who had an urgent need to address their energy security, or the coastal communities of Thailand who pursued the economic benefits of the shrimp industry or urbanization, and the people of Cambodia who worked diligently towards reconstruction after the Khmer rouge. This study allowed us to not only establish a new baseline that would better inform current understandings of mangrove change before the 1990s, but it aided our understanding of the different needs that the people and their governments were trying to meet. We hope that this new baseline and conversation on the political ecology can serve as an example of good research practices in trying to understand mangrove change dynamics without forgetting about the human element.

5 Conclusion

Myanmar, Thailand, and Cambodia are home to one of the planet's most biologically complex and carbon rich mangrove ecosystems. There is a high degree of variability in the cadence and severity of mangrove change in these complex coastal ecosystems. Ultimately, the social and ecological values of these mangrove ecosystems have urged a sustained effort to produce a variety of region and sub-regional mangrove extent data products and inventories for this area. However, mangrove extent and change dynamics before the mid-1990s are not well constrained and this area is no exception. Due to the limited availability of publicly available EO data of sufficient quality or availability, conducting a remote sensing analysis of this nature is supremely difficult. This study therefore worked to identify a semi-automatic approach to quantify mangrove distribution over the course of 1972 to 1977 using the best available Landsat 1-2 MSS Tier 2 data. The extent maps in this study were generated using a Random Forest model that mapped a new baseline extent of

15,421 km^2 with a resulting overall accuracy of 95%. The accuracy assessments also indicated a producer's accuracy of 80% and 98% and a user's accuracy of 90% and 96% for the mangrove and non-mangrove class. The study further established historical losses by comparing the new baseline to external mangrove estimates from GMW. This comparison indicated that mangroves were reduced by 6,830 km^2 (44%) by the year 2020. The majority of mangrove losses therefore occurred between the 1970s and the 1990s followed by an immediate trend in mangrove persistence. This study also elaborated on the political, social, and economic drivers of change for this area and urge the remote sensing community to do the same.

Data availability statement

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

Author contributions

Conceptualization, PB, PM-S, KC, CD, DL, MS, and TF; methodology, PB, CD, PM-S, and TF; validation, PB, PM-S, and CD; formal analysis, PB, PM-S, and CD; data curation, PB, PM-S, and CD; writing—original draft preparation, PB, PM-S, KC, and CD; writing—review and editing, PB, PM-S, KC, CD, DL, TT, MS, and TF; visualization, PB; supervision and project administration, KC, MS, and TF; funding acquisition, KC, DL, MS, and TF. All authors contributed to the article and approved the submitted version.

Funding

All authors were funded by NASA's Land Cover and Land Use Change (LCLUC) Project: "Global Hotspots of Change in Mangrove Forests Project", award number 2020 20-LCLUC2020-0050. Part of this work was performed at the Jet Propulsion Laboratory under contract with the National Aeronautics and Space Administration (NASA).

Acknowledgments

We would like to express our gratitude to folks who helped contribute ideas during the entirety of the project. We would also like to give a special thanks to all the authors who remained committed despite the challenges of an ongoing pandemic or challenges with health.

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.

Publisher's note

All claims expressed in this article are solely those of the authors and do not necessarily represent those of their affiliated organizations, or those of the publisher, the editors and the reviewers. Any product that may be evaluated in this article, or claim that may be made by its manufacturer, is not guaranteed or endorsed by the publisher.

References

Adame, M. F., Connolly, R. M., Turschwell, M. P., Lovelock, C. E., Fatoyinbo, T., Lagomasino, D., et al. (2021). "Future carbon emissions from global mangrove forest loss. *Glob. Chang. Biol.* 27 (12), 2856–2866. doi: 10.1111/gcb.15571

Aksornkoae, S. (2012). Mangroves… coastal treasure of Thailand. *J. R. Institute Thailand* 4, 59–77. Available at: https://www.thaiscience.info/Journals/Article/TJRI/10990648.pdf.

Alban, J. D. T. D., Jamaludin, J., de Wen, D. W., Than, M. M., and Webb, E. L. (2020). Improved estimates of mangrove cover and change reveal catastrophic deforestation in Myanmar. *Environ. Res. Lett.* 15 (3), 0340345. doi: 10.1088/1748-9326/ab666d

Alongi, D. M. (2014). Carbon cycling and storage in mangrove forests. *Annu. Rev. Mar. Sci.* 6, 195–219. doi: 10.1146/annurev-marine-010213-135020

Aung, U.M. (2007). Policy and practice in myanmar's protected area system. *J. Environ. Manage.* 84 (2), 188–2035. doi: 10.1016/j.jenvman.2006.05.016

Bantoon, S. (1994). *Using simulation modelling and remote sensing technique for impact study of shrimp farms on mangrove area and aquatic animal production at welu estuary, khlung district, chantaburi province* (Bangkok: Chulalongkorn University). Unpublished MSc Thesis.

Barbier, E., and Cox, M. (2002). Economic and demographic factors affecting mangrove loss in the coastal provinces of Thailand 1979-1996. *Ambio* 31 (4), 351–575. doi: 10.1579/0044-7447-31.4.351

Belgiu, M., and Drăguţ, L. (2016). Random forest in remote sensing: a review of applications and future directions. *ISPRS J. Photogrammetry Remote Sens.* 114, 24–31. doi: 10.1016/j.isprsjprs.2016.01.011

Bey, A., Sánchez-Paus Díaz, A., Maniatis, D., Marchi, G., Mollicone, D., Ricci, S., et al. (2016). Collect earth: land use and land cover assessment through augmented visual interpretation. *Remote Sens.* 8 (10), 807. doi: 10.3390/rs8100807

Bouillon, S., Borges, A. V., Castañeda-Moya, E., Diele, K., Dittmar, T., Duke, N. C., et al. (2008). Mangrove production and carbon sinks: a revision of global budget estimates. *Global Biogeochem. Cycles* 22 (2), 207. doi: 10.1029/2007GB003052

Boutros-Ghali, B. (1994). *Building peace and development: 1994 report on the work of the organization from the forty-eighth to the forty-ninth session of the general assembly* (New York: United Nations Publications).

Braaten, J. D., Cohen, W. B., and Yang, Z. (2015). Automated cloud and cloud shadow identification in landsat MSS imagery for temperate ecosystems. *Remote Sens. Environ.* 169 (November), 128–138. doi: 10.1016/j.rse.2015.08.006

Brander, L. M., Wagtendonk, A. J., Hussain, S. S., McVittie, A., Verburg, P. H., de Groot, R. S., et al. (2012). Ecosystem service values for mangroves in southeast Asia: a meta-analysis and value transfer application. *Ecosys. Serv.* 1 (1), 62–695. doi: 10.1016/j.ecoser.2012.06.003

Breiman, L. (2001). Random forests. *Mach. Learn.* 45 (1), 5–32. doi: 10.1023/A:1010933404324

Brown, C., Durst, P. B., and Enters, T. (2001). *Forests out of bounds: impacts and effectiveness of logging bans in natural forests in Asia-pacific*. (Bangkok, Thailand: FAO Regional office for Asia and the Pacific).

Bunting, P., Rosenqvist, A., Hilarides, L., Lucas, R. M., Thomas, N., Tadono, T., et al. (2022). Global mangrove extent change 1996–2020: global mangrove watch version 3.0. *Remote Sens.* 14 (15), 36575. doi: 10.3390/rs14153657

Cambodia Forestry Policy Assessment (1996). World Bank, United Nations Development Programme, and Food And Agriculture and Organization.

Cambodia Land Cover Atlas 1985/87–1992/93 (1994). Rome, Italy: Food and agriculture organization.

Chakraborty, S., Sahoo, S., Majumdar, D., Saha, S., and Roy, S. (2019). Future mangrove suitability assessment of Andaman to strengthen sustainable development. *J. Cleaner Product.* 234 (October), 597–614. doi: 10.1016/j.jclepro.2019.06.257

Chamberlain, D. A., Phinn, S. R., and Possingham, H. P. (2021). Mangrove forest cover and phenology with landsat dense time series in central Queensland, Australia. *Remote Sens.* 13 (15), 30325. doi: 10.3390/rs13153032

Charupphat, T., and Charupphat, C. (1997) *Application of landsat-5 (TM) for monitoring the changes of mangrove forest area in Thailand*. Available at: https://scholar.google.com/scholar_lookup?title=Application+of+Landsat-5+%28TM%29+for+monitoring+the+changes+of+mangrove+forest+area+in+Thailand&author=Thongchai+Charupphat&publication_year=1997.

Dahdouh-Guebas, F., and Cannicci, S. (2021). Mangrove restoration under shifted baselines and future uncertainty. *Front. Mar. Sci.* 8. doi: 10.3389/fmars.2021.799543

DeFries, R. S., Rudel, T., Uriarte, M., and Hansen, M. (2010). Deforestation driven by urban population growth and agricultural trade in the twenty-first century. *Nat. Geosci.* 3 (3), 178–815. doi: 10.1038/ngeo756

Devaraj, C., and Shah, C. A. (2014). Automated geometric correction of landsat MSS L1G imagery. *IEEE Geosci. Remote Sens. Lett.* 11 (1), 347–515. doi: 10.1109/LGRS.2013.2257677

Donato, D. C., Kauffman, J.B., Murdiyarso, D., Kurnianto, S., Stidham, M., and Kanninen, M. (2011). Mangroves among the most carbon-rich forests in the tropics. *Nat. Geosci.* 4 (5), 293–975. doi: 10.1038/ngeo1123

Estoque, R. C., Myint, S. W., Wang, C., Ishtiaque, A., Aung, T. T., Emerton, L., et al. (2018). Assessing environmental impacts and change in myanmar's mangrove ecosystem service value due to deforestation, (2000–2014). *Global Change Biol.* 24 (11), 5391–54105. doi: 10.1111/gcb.14409

Everitt, J. H., and Judd, F. W. (1989). Using remote sensing techniques to distinguish and monitor black mangrove (Avicennia germinans). *J. Coast. Res.* 5 (4), 737–745. Available at: https://www.jstor.org/stable/4297605.

Farr, T. G., Rosen, P. A., Caro, E., Crippen, R., Duren, R., Hensley, S., et al. (2007). The shuttle radar topography mission. *Rev. Geophys.* 45 (2). doi: 10.1029/2005RG000183

Faundeen, J. L., Williams, D. L., and Greenhagen, C. A. (2004). Landsat yesterday and today. *J. Map Geogr. Libraries* 1 (1), 59–735. doi: 10.1300/J230v01n01_04

Flood, N. (2013). Seasonal composite landsat TM/ETM+ images using the medoid (a multi-dimensional median). *Remote Sens.* 5 (12), 6481–6500. doi: 10.3390/rs5126481

Food and Agriculture Organization (2007). *The world's mangroves 1980–2005. FAO ForestryPaper 153.*

Friess, D. A., Rogers, K., Lovelock, C. E., Krauss, K. W., Hamilton, S. E., Yip Lee, S., et al. (2019). The state of the world's mangrove forests: past, present, and future. *Annu. Rev. Environ. Res.* 44 (1), 89–1155. doi: 10.1146/annurev-environ-101718-033302

Friess, D. A., Thompson, B. S., Brown, B., Aldrie Amir, A., Cameron, C., Koldewey, H. J., et al. (2016). Policy challenges and approaches for the conservation of mangrove forests in southeast Asia. *Conserv. Biol.* 30 (5), 933–495. doi: 10.1111/cobi.12784

Friess, D. A., and Webb, E. L. (2014). Variability in mangrove change estimates and implications for the assessment of ecosystem service provision. *Global Ecol. Biogeogr.* 23 (7), 715–255. doi: 10.1111/geb.12140

Gandhi, S., and Jones, T. G. (2019). Identifying mangrove deforestation hotspots in south Asia, southeast Asia and Asia-pacific. *Remote Sens.* 11 (6), 7285. doi: 10.3390/rs11060728

Gao, B.-C. (1996). NDWI–a normalized difference water index for remote sensing of vegetation liquid water from space. *Remote Sens. Environ.* 58 (3), 257–266. doi: 10.1016/S0034-4257(96)00067-3

Ghimire, B., Rogan, J., Galiano, V. R., Panday, P., and Neeti, N. (2012). An evaluation of bagging, boosting, and random forests for land-cover classification in cape cod, Massachusetts, USA. *GISci. Remote Sens.* 49 (5), 623–435. doi: 10.2747/1548-1603.49.5.623

Giardino, C., Bresciani, M., Fava, F., Matta, E., Brando, V. E., and Colombo, R. (2016). Mapping submerged habitats and mangroves of lampi island marine national park (Myanmar) from in situ and satellite observations. *Remote Sens.* 8 (1), 25. doi: 10.3390/rs8010002

Gibbs, H. K., Brown, S., Niles, J. O., and Foley, J. A. (2007). Monitoring and estimating tropical forest carbon stocks: making REDD a reality. *Environ. Res. Lett.* 2 (4), 0450235. doi: 10.1088/1748-9326/2/4/045023

Giri, C., Ochieng, E., Tieszen, L. L., Zhu, Z., Singh, A., Loveland, T., et al. (2011). Status and distribution of mangrove forests of the world using earth observation satellite data. *Global Ecol. Biogeogr.* 20 (1), 154–159. doi: 10.1111/j.1466-8238.2010.00584.x

Giri, C., Zhu, Z., Tieszen, L. L., Singh, A., Gillette, S., and Kelmelis, J. A. (2008). Mangrove forest distributions and dynamics, (1975–2005) of the tsunami-affected region of Asia. *J. Biogeogr.* 35 (3), 519–528. doi: 10.1111/j.1365-2699.2007.01806.x

Gitelson, A. A., Viña, A, Arkebauer, T. J., Rundquist, D. C., Keydan, G., and Leavitt, B. (2003). Remote estimation of leaf area index and green leaf biomass in maize canopies. *Geophys. Res. Lett.* 30 (5). doi: 10.1029/2002GL016450

Goldberg, L., Lagomasino, D., Thomas, N., and Fatoyinbo, T. (2020). Global declines in human-driven mangrove loss. *Global Change Biol.* 26 (10), 5844–5555. doi: 10.1111/gcb.15275

Gorelick, N., Hancher, M., Dixon, M., Ilyushchenko, S., Thau, D., and Moore, R. (2017). Google Earth engine: planetary-scale geospatial analysis for everyone. *Remote Sens. Environ.* 202 (December), 18–27. doi: 10.1016/j.rse.2017.06.031

Gupta, K., Mukhopadhyay, A., Giri, S., Chanda, A., Majumdar, S. D., Samanta, S., et al. (2018). An index for discrimination of mangroves from non-mangroves using LANDSAT 8 OLI imagery. *MethodsX* 5, 1129–1139. doi: 10.1016/j.mex.2018.09.011

Hamilton, S. E., and Casey, D. (2016). Creation of a high spatio-temporal resolution global database of continuous mangrove forest cover for the 21st century (CGMFC-21). *Global Ecol. Biogeogr.* 25 (6), 729–385. doi: 10.1111/geb.12449

Hamilton, S. E., Castellanos-Galindo, G. A., Millones-Mayer, M., and Chen, M. (2018). "Remote sensing of mangrove forests: current techniques and existing databases," in *Threats to mangrove forests* (Springer), 497–520. doi: 10.1007/978-3-319-73016-5_22

Hamilton, S. E., and Stankwitz, C. (2012). Examining the relationship between international aid and mangrove deforestation in coastal Ecuador from 1970 to 2006. *J. Land Use Sci.* 7 (2), 177–2025. doi: 10.1080/1747423X.2010.550694

Han, X., Dagsvik, J., and Cheng, Y. (2020). Disability and job constraint in post civil war Cambodia. *J. Dev. Stud.* 56 (12), 2293–23075. doi: 10.1080/00220388.2020.1769073

H. Nguyen, T., Jones, S., Soto-Berelov, M., Haywood, A., and Hislop, S. (2019). Landsat time-series for estimating forest aboveground biomass and its dynamics across space and time: a review. *Remote Sens.* 12 (1), 985. doi: 10.3390/rs12010098

Howard, J., Sutton-Grier, A., Herr, D, Kleypas, J., Landis, E., Mcleod, E., et al. (2017). Clarifying the role of coastal and marine systems in climate mitigation. *Front. Ecol. Environ.* 15 (1), 42–505. doi: 10.1002/fee.1451

Hu, L., Li, W., and Xu, B. (2018). The role of remote sensing on studying mangrove forest extent change. *Int. J. Remote Sens.* 39 (19), 6440–6625. doi: 10.1080/01431161.2018.1455239

Huete, A. R. (1988). A soil-adjusted vegetation index (SAVI). *Remote Sens. Environ.* 25 (3), 295–309. doi: 10.1016/0034-4257(88)90106-X

Islam, Md. M., Borgqvist, H., and Kumar, L. (2019). Monitoring mangrove forest landcover changes in the coastline of Bangladesh from 1976 to 2015. *Geocarto. Int.* 34 (13), 1458–1765. doi: 10.1080/10106049.2018.1489423

Jordan, C. F. (1969). Derivation of leaf-area index from quality of light on the forest floor. *Ecology* 50 (4), 663–666. doi: 10.2307/1936256

Klankamsorn, B., and Charuppat, C. (1982). *Changes of mangrove areas in Thailand by using LANDSAT* Vol. 58 (Bangkok, Thailand: Royal Forest Department Press).

Kodikara, K. A. S., Mukherjee, N., Jayatissa, L. P., Dahdouh-Guebas, F., and Koedam, N. (2017). Have mangrove restoration projects worked? an in-depth study in Sri Lanka. *Restor. Ecol.* 25 (5), 705–165. doi: 10.1111/rec.12492

Kontgis, C., Schneider, A., and Ozdogan, M. (2015). Mapping rice paddy extent and intensification in the Vietnamese Mekong river delta with dense time stacks of landsat data. *Remote Sens. Environ.* 169 (November), 255–269. doi: 10.1016/j.rse.2015.08.004

Kovacs, J. M., de Santiago, F. F., Bastien, J., and Lafrance, P. (2010). An assessment of mangroves in Guinea, West Africa, using a field and remote sensing based approach. *Wetlands* 30 (4), 773–825. doi: 10.1007/s13157-010-0065-3

Kozhikkodan Veettil, B., and Quang, N. X. (2019). Mangrove forests of Cambodia: recent changes and future threats. *Ocean Coast. Manage.* 181 (November), 104895. doi: 10.1016/j.ocecoaman.2019.104895

Kuenzer, C., Bluemel, A., Gebhardt, S., Quoc, T. V., and Dech, S. (2011). Remote sensing of mangrove ecosystems: a review. *Remote Sens.* 3 (5), 878–9285. doi: 10.3390/rs3050878

Kyi, T. M. (1992)Reforestation techniques applied in the ayeyarwady mangroves. In: *Workshop on the conservation and rehabilitation of mangrove resources in Myanmar* (Hmawbi (Myanmar). Available at: https://www.usgs.gov/landsat-missions/landsat-known-issues (Accessed June 27, 2022).

Le Billon, P. (2000). The political ecology of transition in Cambodia 1989–1999: war, peace and forest exploitation. *Dev. Change* 31 (4), 785–805. doi: 10.1111/1467-7660.00177

Lekfuangfu, W. N., and Nakavachara, V. (2021). Reshaping thailand's labor market: the intertwined forces of technology advancements and shifting supply chains. *Econ. Model.* 102 (September), 105561. doi: 10.1016/j.econmod.2021.105561

Lewis, A. J., and MacDonald, H. C. (1972). Mapping of mangrove and perpendicular-oriented shell reefs in southeastern Panama with side-looking radar. *Photogrammetria* 28 (6), 187–995. doi: 10.1016/0031-8663(72)90001-4

Li, P., Feng, Z., and Xiao, C. (2018). Acquisition probability differences in cloud coverage of the available landsat observations over mainland southeast Asia from 1986 to 2015. *Int. J. Digital Earth* 11 (5), 437–505. doi: 10.1080/17538947.2017.1327619

Li, M. S., Mao, L. J., Shen, W. J., Liu, S. Q., and Wei, A. S. (2013). "Change and fragmentation trends of zhanjiang mangrove forests in southern China using multi-temporal landsat imagery, (1977–2010)," in *Estuarine, coastal and shelf science,* *pressures, stresses, shocks and trends in estuarine ecosystems*. (London: Elsevier Ltd), 130, 111–120. doi: 10.1016/j.ecss.2013.03.023

Lim, C. L., Prescott, G. W., De Alban, J. D. T., Ziegler, A. D., and Webb, E. L. (2017). Untangling the proximate causes and underlying drivers of deforestation and forest degradation in Myanmar. *Conserv. Biol.: J. Soc. Conserv. Biol.* 31 (6), 1362–1725. doi: 10.1111/cobi.12984

Lisle, R. J. (2006). Google Earth: a new geological resource. *Geol. Today* 22 (1), 29–32. doi: 10.1111/j.1365-2451.2006.00546.x

Locard, H. (2005). State violence in democratic Kampuchea, (1975–1979) and retribution, (1979–2004). *Eur. Rev. History: Rev. Européenne d'histoire* 12 (1), 121–143. doi: 10.1080/13507480500047811

Lorenzo, E. N., de Jesus, B. R.Jr., and Jara, R. S. (1979). "Assessment of mangrove forest deterioration in zamboanga peninsula, Philippines using landsat MSS data," in *Proceedings of the Thirteenth International Symposium on Remote Sensing of Environment*, Ann Arbor, Mich, 23-27 April 1979 (Environmental Research Institute of Michigan).

Loveland, T. R., and Dwyer, J. L. (2012). Landsat: building a strong future. *Remote Sens. Environ.* 122 (July), 22–29. doi: 10.1016/j.rse.2011.09.022

Mcowen, C. J., Weatherdon, L. V., Bochove, J.-W. V., Sullivan, E., Blyth, S., Zockler, C., et al. (2017). A global map of saltmarshes. *Biodiversity Data J.* 5 (March), e11764. doi: 10.3897/BDJ.5.e11764

Ministry of Environment, Kingdom of Cambodia (2009). *Cambodia environment outlook* (Phnom Penh, Cambodia: Ministry of Environment). Available at: https://www.unep.org/resources/report/cambodia-environmental-outlook.

Mosyakov, D. (2000). The Khmer rouge and the Vietnamese communists: a history of their relations as told in the soviet archives. *Cambodian Genocide Project Yale Univ*. Available at: https://www.files.ethz.ch/isn/46645/GS20.pdf.

Murillo-Sandoval, P. J., Fatoyinbo, L., and Simard, M. (2022). Mangroves cover change trajectories 1984-2020: the gradual decrease of mangroves in Colombia. *Front. Mar. Sci.* 9. doi: 10.3389/fmars.2022.892946

Murray, N. J., Worthington, T. A., Bunting, P., Duce, S., Hagger, V., Lovelock, C. E., et al. (2022). High-resolution mapping of losses and gains of earth's tidal wetlands. *Science* 376 (6594), 744–749. doi: 10.1126/science.abm9583

Naito, T., and Traesupap, S. (2014). "The relationship between mangrove deforestation and economic development in Thailand," in *Mangrove ecosystems of Asia: status, challenges and management strategies*. Eds. I. Faridah-Hanum, A. Latiff, K. R. Hakeem and M. Ozturk (New York, NY: Springer), 273–294. doi: 10.1007/978-1-4614-8582-7_13

Nop, S., DasGupta, R., and Shaw, R. (2017). Opportunities and challenges for participatory management of mangrove resource (PMMR) in Cambodia. *Participatory Mangrove Manage. Changing Climate*, 187–202. doi: 10.1007/978-4-431-56481-2_12

Oliphant, A. J., Thenkabail, P. S., Teluguntla, P., Xiong, J., Gumma, M. K., Congalton, R. G., et al. (2019). Mapping cropland extent of southeast and northeast Asia using multi-year time-series landsat 30-m data using a random forest classifier on the Google earth engine cloud. *Int. J. Appl. Earth Observation Geoinf.* 81 (September), 110–124. doi: 10.1016/j.jag.2018.11.014

Olofsson, P., Foody, G. M., Herold, M., Stehman, S. V., Woodcock, C. E., and Wulder, M. A. (2014). Good practices for estimating area and assessing accuracy of land change. *Remote Sens. Environ.* 148, 42–57. doi: 10.1016/j.rse.2014.02.015

Olofsson, P., Foody, G. M., Stehman, S. V., and Woodcock, C. E. (2013). Making better use of accuracy data in land change studies: estimating accuracy and area and quantifying uncertainty using stratified estimation. *Remote Sens. Environ.* 129, 122–131. doi: 10.1016/j.rse.2012.10.031

Oo, N. (2002). Present state and problems of mangrove management in Myanmar. *Trees* 16 (2), 218–223. doi: 10.1007/s00468-001-0150-6

Otsu, N. (1979). A threshold selection method from Gray-level histograms. *IEEE Trans. Sys. Man Cybernetics* 9 (1), 62–66. doi: 10.1109/TSMC.1979.4310076

Pal, M. (2003). "Random forests for land cover classification," in *IGARSS 2003. 2003 IEEE International Geoscience and Remote Sensing Symposium. Proceedings (IEEE Cat. No. 03CH37477)*, Vol. 6. 3510–3512 (Toulouse France: IEEE).

Pasquarella, V. J., Holden, C. E., Kaufman, L., and Woodcock, C. E. (2016). From imagery to ecology: leveraging time series of all available landsat observations to map and monitor ecosystem state and dynamics. *Remote Sens. Ecol. Conserv.* 2 (3), 152–705. doi: 10.1002/rse2.24

Pednekar, S. S. (1998). *Background report for the Thai marine rehabilitation plan 1997-2001*. (Bangkok, Thailand: Thailand Development Research Institute Foundation).

Peel, M. C., Finlayson, B. L., and McMahon, T. A. (2007). Updated world map of the köppen-Geiger climate classification. *Hydrol. Earth Sys. Sci.* 11 (5), 1633–1445. doi: 10.5194/hess-11-1633-2007

Polidoro, B. A., Carpenter, K. E., Collins, L., Duke, N. C., Ellison, A. M., Ellison, J. C., et al (2010). The loss of species: mangrove extinction risk and geographic areas of global concern. *PloS One* 5 (4), e10095. doi: 10.1371/journal.pone.0010095

Pumijumnong, N. (2014). "Mangrove forests in Thailand," in *Mangrove ecosystems of Asia: status, challenges and management strategies*. Eds. I. Faridah-Hanum, A. Latiff, K. R. Hakeem and M. Ozturk (New York, NY: Springer), 61–79. doi: 10.1007/978-1-4614-8582-7_4

Rahman, M.M. (2012). Time-series analysis of coastal erosion in the sundarbans mangrove. *Int. Arch. Photogrammetry Remote Sens. Spatial Inf. Sci.* 39 (B8), 425–429. doi: 10.5194/isprsarchives-XXXIX-B8-425-2012

Reddy, C.S., Pattanaik, C., and Murthy, M. S. R. (2007). Assessment and monitoring of mangroves of bhitarkanika wildlife sanctuary, orissa, India using remote sensing and GIS. *Curr. Sci.* 92 (10), 1409–1415.

Richards, D. R., and Friess, D. A. (2016). Rates and drivers of mangrove deforestation in southeast Asia 2000–2012. *Proc. Natl. Acad. Sci.* 113 (2), 344–495. doi: 10.1073/pnas.1510272113

Richards, D. R., Thompson, B. S., and Wijedasa, L. (2020). Quantifying net loss of global mangrove carbon stocks from 20 years of land cover change. *Nat. Commun.* 11 (1), 42605. doi: 10.1038/s41467-020-18118-z

Rizvi, A. R., and Singer, U. (2016). *Cambodia Coastal situation analysis* (Switzerland: International Union for Conservation of Nature). Available at: https://www.iucn.org/content/cambodia-coastal-situation-analysis.

Rondeaux, G., Steven, M., and Baret, F. (1996). Optimization of soil-adjusted vegetation indices. *Remote Sens. Environ.* 55 (2), 95–1075. doi: 10.1016/0034-4257(95)00186-7

Roy, D. P., Kovalskyy, V., Zhang, H. K., Vermote, E. F., Yan, L., Kumar, S. S., et al. (2016). Characterization of landsat-7 to landsat-8 reflective wavelength and normalized difference vegetation index continuity. *Remote Sens. Environ.* 185, 57–70. doi: 10.1016/j.rse.2015.12.024

Ruiz-Luna, A., Acosta-Velázquez, J., and Berlanga-Robles, C. (2008). On the reliability of the data of the extent of mangroves: a case study in Mexico. *Ocean Coast. Manage.* 51 (4), 342–515. doi: 10.1016/j.ocecoaman.2007.08.004

Saah, D., Johnson, G., Ashmall, B., Tondapu, G., Tenneson, K., Patterson, M., et al. (2019). Collect earth: an online tool for systematic reference data collection in land cover and use applications. *Environ. Model. Softw.* 118 (August), 166–171. doi: 10.1016/j.envsoft.2019.05.004

Sayre, R., Noble, S., Hamann, S., Smith, R., Wright, D., Breyer, S., et al. (2019). A new 30 meter resolution global shoreline vector and associated global islands database for the development of standardized ecological coastal units. *J. Operational Oceanogr.* 12 (sup2), S47–S56. doi: 10.1080/1755876X.2018.1529714

Serra, J. (1982). Image analysis and mathematical morphol-ogy. vol. 1–2.

Simard, M., Fatoyinbo, L., Smetanka, C., Rivera-Monroy, V. H., Castañeda-Moya, E., Thomas, N., et al. (2019). Mangrove canopy height globally related to precipitation, temperature and cyclone frequency. *Nat. Geosci.* 12 (1), 40–455. doi: 10.1038/s41561-018-0279-1

Singh, G., Ramanathan, A. L., and Prasad, M. BALA KRISHNA (2005). Nutrient cycling in mangrove ecosystem: a brief overview. *Int. J. Ecol. Environ. Sci.* 30, 231–244.

Son, N.-T., Chen, C.-F., Chang, N.-B., Chen, C.-R., Chang, L.-Y., and Thanh, B.-X. (2014). Mangrove mapping and change detection in Ca mau peninsula, Vietnam, using landsat data and object-based image analysis. *IEEE J. Sel. Topics Appl. Earth Observations Remote Sens.* 8 (2), 503–105. doi: 10.1109/JSTARS.2014.2360691

Spalding, M., Kainuma, M., and Collins, L. (2010). *World atlas of mangroves* (London, UK: Washington, DC: Routledge).

Takaku, J., Tadono, T., Doutsu, M., Ohgushi, F., and Kai, H. (2020). UPDATES OF'AW3D30'ALOS GLOBAL DIGITAL SURFACE MODEL WITH OTHER OPEN ACCESS DATASETS. *Int. Arch. Photogrammetry Remote Sens. Spatial Inf. Sci.* 43, 183–189. doi: 10.5194/isprs-archives-XLIII-B4-2020-183-2020

Takaku, J., Tadono, T., and Tsutsui, K. (2014). GENERATION OF HIGH RESOLUTION GLOBAL DSM FROM ALOS PRISM. *ISPRS Ann. Photogrammetry Remote Sens. Spatial Inf. Sci.* 2 (4), 243–248. doi: 10.5194/isprsarchives-XL-4-243-2014

Tarpley, J. D., Schneider, S. R., and Money, R. L. (1984). Global vegetation indices from the NOAA-7 meteorological satellite. *J. Climate Appl. Meteorol.* 23 (3), 491–494. doi: 10.1175/1520-0450(1984)023<0491:GVIFTN>2.0.CO;2

Thant, Y. M., Kanzaki, M., Ohta, S., and Than, M. M. (2012). Carbon sequestration by mangrove plantations and a natural regeneration stand in the ayeyarwady delta, Myanmar. *Tropics* 21 (1), 1–105. doi: 10.3759/tropics.21.1

Thomas, N., Lucas, R., Bunting, P., Hardy, A., Rosenqvist, A., and Simard, M. (2017). Distribution and drivers of global mangrove forest change 1996–2010. *PloS One* 12 (6), e01793025. doi: 10.1371/journal.pone.0179302

U.S. Geological Survey, Department of the Interior (2018) *Landsat multispectral scanner (MSS) level 1 (L1) data format control book (DFCB)." U.S. geological survey*. Available at: https://earth.esa.int/eogateway/documents/20142/37627/LSDS-286_LandsatMSS-Level1_DFCB-v11.pdf.

Valiela, I., Bowen, J. L., and York, J. K. (2001). Mangrove forests: one of the world's threatened major tropical environments: At least 35% of the area of mangrove forests has been lost in the past two decades, losses that exceed those for tropical rain forests and coral reefs, two other well-known threatened environments. *Bioscience* 51 (10), 807–155. doi: 10.1641/0006-3568(2001)051[0807:MFOOTW]2.0.CO;2

Veettil, B. K., Pereira, S. F. R., and Quang, N. X. (2018). Rapidly diminishing mangrove forests in Myanmar (Burma): a review. *Hydrobiologia* 822 (1), 19–355. doi: 10.1007/s10750-018-3673-1

Vogeler, J. C., Braaten, J. D., Slesak, R. A., and Falkowski, M. J. (2018). Extracting the full value of the landsat archive: inter-sensor harmonization for the mapping of Minnesota forest canopy cover, (1973–2015). *Remote Sens. Environ.* 209 (May), 363–374. doi: 10.1016/j.rse.2018.02.046

Wang, L., Jia, M., Yin, D., and Tian, J. (2019). A review of remote sensing for mangrove forests: 1956–2018. *Remote Sens. Environ.* 231 (September), 111223. doi: 10.1016/j.rse.2019.111223

Webb, E. L., Jachowski, N. R.A., Phelps, J., Friess, D. A., Than, M. M., and Ziegler, A. D. (2014). Deforestation in the ayeyarwady delta and the conservation implications of an internationally-engaged Myanmar. *Global Environ. Change* 24 (January), 321–333. doi: 10.1016/j.gloenvcha.2013.10.007

Worthington, T., and Spalding, M. (2018). *Mangrove restoration potential: a global map highlighting a critical opportunity*. Apollo - University of Cambridge Repository. 3–29. doi: 10.17863/CAM.39153

Wulder, M. A., Roy, D. P., Radeloff, V. C., Loveland, T. R., Anderson, M. C., Johnson, D. M., et al. (2022). Fifty years of landsat science and impacts. *Remote Sens. Environ.* 280 (October), 113195. doi: 10.1016/j.rse.2022.113195

Wulder, M. A., White, J. C., Loveland, T. R., Woodcock, C. E., Belward, A. S., Cohen, W. B., et al. (2016). The global landsat archive: status, consolidation, and direction. *Remote Sens. Environ.* 185 (November), 271–283. doi: 10.1016/j.rse.2015.11.032

Yan, L., and Roy, D. P. (2021). Improving landsat multispectral scanner (MSS) geolocation by least-Squares-Adjustment based time-series Co-registration. *Remote Sens. Environ.* 252 (January), 112181. doi: 10.1016/j.rse.2020.112181

Yancho, J., Maxwell, M., Jones, T. G., Gandhi, S. R., Ferster, C., Lin, A., et al. (2020). The Google earth engine mangrove mapping methodology (GEEMMM). *Remote Sens.* 12 (22), 37585. doi: 10.3390/rs12223758

Yang, L., Meng, X., and Zhang, X. (2011). SRTM DEM and its application advances. *Int. J. Remote Sens.* 32 (14), 3875–3965. doi: 10.1080/01431161003786016

Zhu, Z., Qiu, S., He, B., and Deng, C. (2018). Cloud and cloud shadow detection for landsat images: the fundamental basis for analyzing landsat time series. *Remote Sens. Time Ser. Image Process.*, 3–23. doi: 10.1201/9781315166636-1

Zhu, Z., Wulder, M. A., Roy, D. P., Woodcock, C. E., Hansen, M. C., Radeloff, V. C., et al. (2019). Benefits of the free and open landsat data policy. *Remote Sens. Environ.* 224, 382–385. doi: 10.1016/j.rse.2019.02.016

Zöckler, C., and Aung, C. (2019). The mangroves of myanmar. *Sabkha Ecosystems*: Volume VI: Asia/Pacific, pp. 253–268.

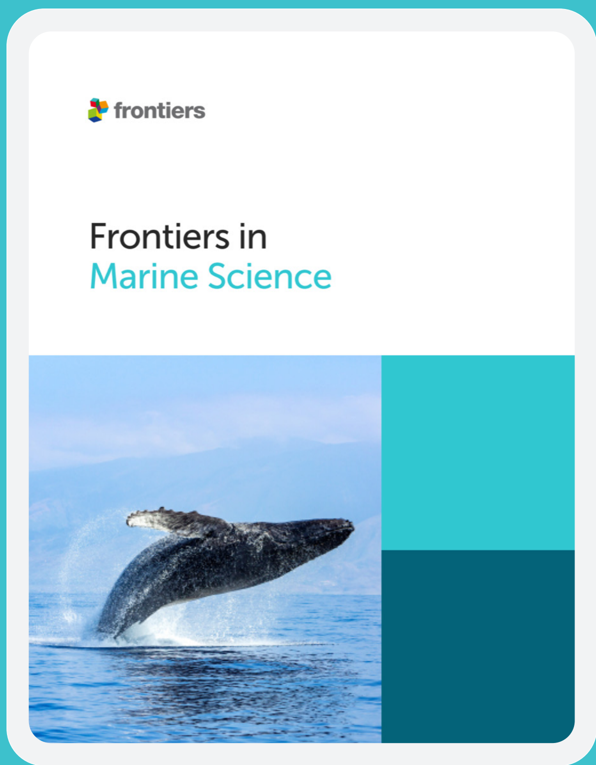
Frontiers in
Marine Science

Explores ocean-based solutions for emerging global challenges

The third most-cited marine and freshwater biology journal, advancing our understanding of marine systems and addressing global challenges including overfishing, pollution, and climate change.

Discover the latest Research Topics

See more →



Frontiers

Avenue du Tribunal-Fédéral 34
1005 Lausanne, Switzerland
frontiersin.org

Contact us

+41 (0)21 510 17 00
frontiersin.org/about/contact