Estuarine, Coastal and Shelf Science 259 (2021) 107467

Contents lists available at ScienceDirect

Estuarine, Coastal and Shelf Science

journal homepage: www.elsevier.com/locate/ecss

Effects of diverse mangrove management practices on forest structure, carbon dynamics and sedimentation in North Sumatra, Indonesia

Bayu B. Hanggara [a], Daniel Murdiyarso [a,b,*], Yohanes RS. Ginting [b], Yessica L. Widha [b], Grace Y. Panjaitan [c], Ali A. Lubis [d]

[a] Center for International Forestry Research (CIFOR), Bogor, 16115, Indonesia
[b] Department of Geophysics and Meteorology, IPB University, Bogor, 16680, Indonesia
[c] Yayasan Gajah Sumatra (YAGASU), Medan, 20153, Indonesia
[d] Center for Isotopes and Radiation Application, National Nuclear Energy Agency (BATAN), Jl. Lebak Bulus Raya 49, Jakarta, 12440, Indonesia

ARTICLE INFO

Keywords:
Biodiversity
Blue carbon
Carbon stocks
Environmental isotope ^{210}Pb
rSET-MH

ABSTRACT

For decades, mangrove forests have been under tremendous pressure due to deforestation and conversion. To sustainably manage the mangroves that remain, an ecosystem approach to management is essential. Two different management regimes – conservation and restoration – were assessed, looking at their respective effects on forest structure and carbon cycling capacity, when compared with degraded mangrove. We found that mangrove restoration enhanced tree density, while mangrove conservation was able to maintain species diversity. In terms of carbon budgets, aboveground carbon was lower in restored mangrove (79.40 ± 37.41 Mg C ha^{-1}) when compared with conserved mangrove (92.26 ± 22.65 Mg C ha^{-1}), but was almost double that found in degraded mangrove (39.89 ± 27.49 Mg C ha^{-1}). Although conserved mangrove had higher aboveground carbon, lower amounts of soil carbon were found in conserved mangrove (127.49 ± 33.21 Mg C ha^{-1}) than in restored and degraded mangrove (236.26 ± 20.33 Mg C ha^{-1} and 139.17 ± 25.44 Mg C ha^{-1}, respectively). The elevation change was highest in degraded mangrove (41.7 ± 24.0 mm yr^{-1}), followed by restored (20.7 ± 14.6 mm yr^{-1}) and conserved mangrove (12.2 ± 3.9 mm yr^{-1}). Carbon burial in conserved mangrove (1.20 ± 1.90 Mg C ha^{-2} yr^{-1}) was double that of degraded mangrove (0.63 ± 0.60 Mg C ha^{-2} yr^{-1}). Ultimately, we conclude that although a conserved mangrove is not always the end result of mangrove restoration and sustainable management, finding balance between structural development and ecosystem function is essential to serve different objectives, including biodiversity maintenance.

1. Introduction

Mangroves provide a diverse range of valuable ecosystem services, which include supporting services (e.g. nutrient cycling, net primary production and land formation), provisioning services (e.g. food, fuel and fiber products), and regulating services (e.g. protection from climate change, flooding, storm surges and pollution, and water purification) (Millenium Ecosystem Assessment, 2005). Together with seagrasses and salt marshes, mangrove forests represent the 'blue carbon' ecosystem, which has incredible biomass productivity and provides efficient long-term carbon sinks (Bouillon et al., 2008; Lovelock and Duarte, 2019).

However, mangrove ecosystems across the world are under pressure from deforestation, degradation, and conversion. In Southeast Asia, the rate of loss of mangrove forests between 2000 and 2012 was reported as 0.18% per year (Richards and Friess, 2016). While, in Indonesia, mangrove area has declined at a rate of 1.24% per year, from 4.2 Mha in 1980 to 2.9 Mha in 2005 (FAO, 2007). The last estimates indicate that Indonesia's remaining mangrove area is approximately 3.1 Mha (Giri et al., 2011).

The impacts of deforestation and degradation significantly reduce the carbon sink capacity of mangroves. Although mangrove loss counted for just 6% of Indonesia's total annual deforestation (Margono et al., 2014), this loss contributed to annual greenhouse gas (GHG) emissions of 0.07–0.21 Pg CO_2e, equivalent to 10–31% of the total estimate for emissions from land-use sectors (Murdiyarso et al., 2015). Avoiding mangrove deforestation could make a significant contribution to reducing GHG emissions; mangroves hold as much as 1083 ± 378 Mg C

* **Corresponding author**. Center for International Forestry Research (CIFOR), Bogor, 16115, Indonesia.
E-mail address: d.murdiyarso@cgiar.org (D. Murdiyarso).

https://doi.org/10.1016/j.ecss.2021.107467
Received 22 September 2020; Received in revised form 17 May 2021; Accepted 7 June 2021
Available online 12 June 2021
0272-7714/

ha^{-1} of total ecosystem carbon stocks, which equates to carbon storage of 3.14 ± 1.10 Pg C (Murdiyarso et al., 2015).

Maintaining mangrove intact would equally allow the forest to enhance any carbon inputs that arise from organic carbon burial or other sources. Organic carbon that enters the mangrove ecosystem originates from two interrelated sources, allochthonous (e.g. tidal waves bringing marine inputs as well as fluvial sediment from upstream) and autochthonous (i.e. on-site biomass carbon input) (Adame et al., 2010). Globally, organic carbon burial rates range from 0.2 to 10.2 Mg C ha^{-1} yr^{-1} (Breithaupt et al., 2012). Organic carbon burial rates in Indonesia have been shown to be as much as 6.58 Mg C ha^{-1} yr^{-1} in Central Java (Kusumaningtyas et al., 2019), up to 2.20 Mg C ha^{-1} yr^{-1} in Bali (Sidik et al., 2019), around 17.22 Mg C ha^{-1} yr^{-1} in East Kalimantan (Kusumaningtyas et al., 2019), and up to 1.19 Mg C ha^{-1} yr^{-1} in West Papua (Sasmito et al., 2020a).

The connectivity between forest structure, sedimentation and carbon burial in mangrove ecosystems has been reported, particularly because improvements to forest structure would facilitate carbon accumulation through sedimentation processes. A study conducted in Sydney Harbor mangrove showed that young mangrove tree roots recolonized and then facilitated soil volume expansion and mineral sedimentation to yield a small soil surface elevation increment of 2.9 mm yr^{-1} (Rogers et al., 2005). In addition, increased mangrove density promoted an increase to surface elevation in various ways, such as vertical accretion, retention of deposited sediments, and root growth (Huxham et al., 2010).

Naturally, mangroves have self-recovering abilities if their morphological and hydrological features have not been changed or damaged (Martinuzzi et al., 2009). However, when disturbances occur, these abilities slowly degrade and/or disappear. A recent study in West Papua, Indonesia, indicated that aboveground biomass requires a minimum of 25 years to recover and reach the same levels of biomass carbon seen in undisturbed mangrove forest (Sasmito et al., 2020b). It is not known, however, to what extent human-induced restoration or rehabilitation efforts could enhance this recovery process, in terms of forest structure and functioning.

Restoring degraded mangrove has the potential to improve the functioning of coastal forests and carbon stocks, but ensuring species diversity is often neglected. It is essential that the climate mitigation agenda, through mangrove restoration, aims at reversing biodiversity losses (Nellemann et al., 2009). Depending on the availability of plant materials, human-induced restoration has had a tendency to use mono-species, such as that found in the severely degraded mangroves of Segara Anakan Lagoon, Central Java, where *Rhizophora apiculata* was planted in the mudflat zone and *Bruguiera gymnorrhiza* was planted in harder substrate in the fringing and interior zones (Soemodihardjo et al., 1991). Although non-planted species have the chance to colonize the restored area, they are usually suppressed by any planted mono-species, as seen in Gazi Bay, Kenya (Kairo et al., 2001). This type of management practice does not encourage the natural regeneration that permits the retrieval of multiple species, as found in Cimanuk River Delta in West Java (Sukardjo et al., 2014).

The purpose of this study was to understand the best management practice for sustainable mangrove management. We assessed a number of biophysical parameters related to forest structure, carbon dynamics and sedimentation processes, looking at mangroves under different management regimes (i.e. conservation and restoration), with the primary objective of comparing the results against mangroves in degraded conditions, which happen to dominate the coastal landscape in North Sumatra, Indonesia. We hypothesize that good management practices will improve forest structure and carbon stocks, as well as the sedimentation and carbon burial rates. It is expected that study findings can inform public policy making processes, particularly regarding restoration efforts and the use of blue carbon mangrove ecosystems as part of nature-based climate solutions.

2. Methods

2.1. Study sites

Study sites were established in the regencies of Langkat and Deli Serdang in North Sumatra Province, Indonesia. Three different mangrove stands were selected: Jaring Halus (JH) to represent mangrove under conservation; Belawan (B) to represent restored mangrove (30 years after restoration); and Percut (P) to represent degraded mangrove. All studied mangrove areas were located between 3°43′ and 3°56′ N and between 98°33′ and 98°47′ E (Fig. 1). The mangrove forests are subject to a monsoonal climate, characterized by a mean temperature of 30 °C with annual rainfall of 1848 mm (BPSNorth Sumatra, 2015). Floristically, the area is dominated by the genera of *Rhizophoraceae, Meliaceae* and *Avicenniaceae*.

The site of Jaring Halus (3°56′ 30.46″ N and 98°34′ 8.9" E) is a small island of 90 ha and forms part of the Karang Gading and Langkat Timur Laut wildlife reserves on the mainland. Although the eastern part of the island is occupied by a fishing community, the local people co-exist with the mangrove conservation area. The village head is very influential in implementing customary law to keep the mangrove forests intact. As shown in Fig. 1, this site is characterized by its estuarine setting. The soil texture in the interior is dominated by clay, with silt gradually occupying an unstable fringe and dominating the labile mudflat. This conserved mangrove shows low disturbance, and the mangroves are largely intact.

The Belawan site (3°46′ 4.82″ N and 98°42′ 44.43" E) has riverine characteristics with a narrow fringe and an absence of mudflats. This restored mangrove covers an area of 104 ha and lies approximately 19.6 km northeast of Medan, the capital city of North Sumatra, with its population of 2.23 million (BPSNorth Sumatra, 2015). It is adjacent to a large seaport, container yards and an international airport. The study site is directly affected by waves from commercial vessels traffic at Belawan Seaport, urban solid waste, and pollution. Before 1980, this area was made up of active ponds; however, after losing productivity, these fishponds were abandoned, and mangroves slowly regenerated naturally. Supported by a local non-government organization, the local community subsequently enhanced this restoration by replanting *Avicennia* sp. and *Rhizophora* sp. in the abandoned ponds, which are predominantly clay in texture. Since then, they have patrolled the area to protect it from illegal logging.

In contrast, Percut is a degraded mangrove forest which undergoes no management activity. The area, which is located at 3°43′ 31.2″ N and 98°47′ 22.6" E, has deltaic characteristics with rich sediment input from the catchment area. This mangrove forest spans approximately 147 ha and is bordered by two channels. The interior is dominated by abandoned fishponds and is connected to settlements, whose communities extract timber to supply local charcoal kilns or for light building material. Percut mangrove, which grows on predominantly clay soil, is also threatened by adjacent oil palm expansion.

2.2. Mangrove forest structure

Data collection in Belawan and Percut mangroves was conducted in February 2016, while in Jaring Halus mangrove, data collection took place in December 2017. The establishment of transects or plots, consisting of six sub-plots, followed a commonly-used protocol (Kauffman and Donato, 2012). Five transects were established in Jaring Halus (JH1-JH5), three transects in each Belawan site (B1–B3), and three transects were established in Percut (P1–P3), to enable forest structure analysis and carbon assessments.

We measured Diameter at Breast Height (DBH) and identified the species of trees (DBH >5 cm) within 7 m radius sub-plots, as well as identifying the species of saplings (DBH <5 cm) and seedlings within 2 m radius concentric sub-plots. From these measurements we derived tree density and basal area (BA).

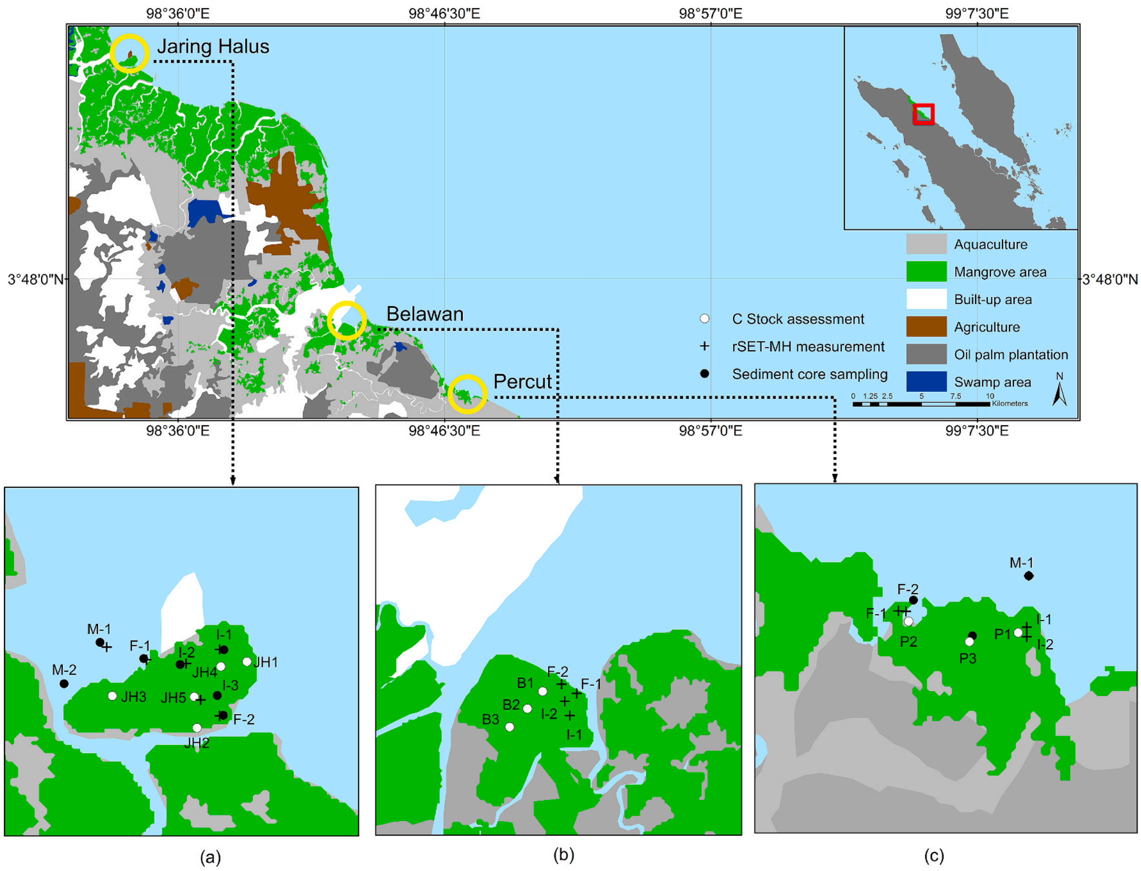

Fig. 1. Map of the study sites on the east coast of North Sumatra, Indonesia, consisting of: (a) conserved mangrove, Jaring Halus (JH); (b) restored mangrove, Belawan (B); and (c) degraded mangrove, Percut (P). Measurements of carbon stocks (○), monitoring of contemporary sedimentation using rSET-MH (+), and sediment core sampling (●) were unevenly distributed across the Mudflats (M), Fringe (F), and Interior (I) mangroves.

Species richness and diversity metrics (i.e. Shannon Diversity Index, H' (Shannon and Weaver, 1963), Simpson diversity index, D (Simpson, 1949), and Pielou evenness Index, J' (Pielou, 1966)) were calculated for each site. Diversity indices measure the degree of diversity when individuals within a population are classified into groups of species; H' ranges from 0 to 5, and D ranges from 0 to 1. The evenness index measures how evenly the number of species are distributed across diversity indices; numbers close to 0 mean that only a few species are dominant and driving diversity. The importance value index (IVI) of each species was estimated by accumulating the values of relative frequency, relative density, and relative dominance, measured for each plot at the three sites (Cintron and Schaeffer-Novelli, 1984; Datta and Deb, 2017).

2.3. Total ecosystem carbon stocks assessment

Aboveground biomass (AGB) and belowground biomass (BGB) per unit area (Mg ha^{-1}) were estimated through species-specific allometric equations (see Table S1). Aboveground carbon (AGC) stocks per unit area (Mg C ha^{-1}) were calculated using a carbon conversion factor of 0.47; for belowground carbon (BGC) stocks, including belowground roots, stocks were calculated using 0.39 as a conversion factor.

Following Kauffman and Donato (2012), dead wood debris (DWD) was estimated through collection of downed deadwood or woody debris using the planar intercept technique. Downed deadwood was classed into four diameters (D): fine ($D < 0.6$ cm), small ($0.6 \text{ cm} < D < 2.5$ cm), medium ($2.5 \text{ cm} < D < 7.5$ cm), and large – sound or rotten ($D > 7.5$ cm).

Soil organic carbon (SOC) stocks were estimated from soil cores collected in each sub-plot using a 5.5 cm diameter open-faced auger. Approximately 5 cm of sediment was extracted using consistent depth intervals: 0 to 15, 15 to 30, 30 to 50, and 50–100 cm; and consistently taken at the middle interval. Soil samples were dried at 40 °C until a constant mass was reached. Samples were ground using a pestle and mortar then sieved with a 0.5 mm mesh to remove large roots and inorganic debris. The soil carbon content was then quantified used a dry combustion technique, utilizing a *LECO* CNS elemental analyzer.

Total ecosystem carbon stocks (TECS) are the sum of AGC, BGC, DWD and SOC in transects from fringe and interior mangroves. In the mudflat, only SOC is used to represent total ecosystem carbon stocks.

2.4. Contemporary sedimentation and carbon burial rates

Contemporary sedimentation was monitored using rod Surface Elevation Table (rSET) and Marker Horizon (MH) instruments (Cahoon et al., 2002). Hydrogeomorphology was taken into consideration when installing the rSET-MH in the Mudflat (M), Fringe (F) (~15 m from the ocean), and Interior (I) (~375 m from the ocean) locations (MacKenzie et al., 2016). As shown in Fig. 1, six rSET-MH instruments were installed in Jaring Halus (one in the mudflat, M-1; two in the fringe mangrove, F-1 and F-2; and three in the interior mangrove, I-1, I-2 and I-3). Four were installed in Belawan (two in each fringe and interior mangrove, F-1, F-2, I-1 and I-2). While, five rSET-MH instruments were installed in Percut (one in the mudflat, M-1; two in the fringe mangrove, F-1 and F-2; and two in the interior mangrove, I-1 and I-2). A baseline measurement was taken three months after the instruments were first installed to allow the stations to stabilize. Data were collected annually to monitor surface elevation change (SEC) using the rSET, and surface accretion rates (SAR) using MH instruments.

In this study, we used rSET instruments with nine pins in the rSET arm and measured in four different arm directions (i.e. a total of 36 SEC measurements were taken at each station). At the time of data collection, any natural disturbances (e.g. tree roots, pneumatophores, crab mounds, footprints) that interfered with the true ground surface, where noted, were omitted during analysis, along with their associated measurements.

SAR or vertical sediment accretion was measured directly using MH instruments (Cahoon et al., 1995; Lynch et al., 2015; Bomer et al., 2020). In this study, for MH instruments, we employed a 25 cm long x 25 cm wide square stainless-steel mesh, with a thickness of 0.5 cm and a mesh size of 1 cm diameter. We installed two MH instruments next to each rSET station, in diagonal directions. We put MH instruments in the soil surface and secured around the edges with pins to make sure they were stable and safe. Accretion was measured by looking at how much soil had accumulated above the mesh each time. On each occasion, we recorded five separate values of soil accretion for each MH, using randomization sampling. The average measurement across the two MH instruments ($n = 10$) was then calculated to obtain the periodic accretion of each rSET-MH station.

The difference between SAR and SEC was used to estimate shallow subsidence (SS) (Cahoon et al., 1995). Positive values indicate that SS has occurred at a depth between the ground surface and the hard layer, while negative values show that shallow sub-surface expansion has occurred.

Carbon burial rates were estimated by multiplying the accretion rates by carbon concentration and bulk density (Marchio et al., 2016). As carbon concentration and bulk density are not measured in accreted sediment, we made the assumption that they were the same as the values obtained from the ^{210}Pb cores (Breithaupt et al., 2020). At the Belawan site, we used carbon density from the first layer of soil carbon cores, due to the absence of ^{210}Pb sampling.

2.5. Historical sedimentation and carbon burial rates

Historical sediment data were obtained using the radionuclide technique of dating. Sediment cores intended for this purpose were separated from soil cores for soil carbon stock assessments. Two cores were collected from each hydrogeomorphic setting in the Jaring Halus conserved mangroves (M-1, M-2, F-1, F-2, I-1, I-2), and one core was collected from each hydrogeomorphic setting in the Percut degraded mangroves (M-1, F-1, I-1), from areas close to the rSET-MH stations (see Fig. 1).

Sediment cores were extracted using a stainless steel Eijkelkamp peat auger (volume = 67.7 cm^3). The sediment cores were collected up to a depth of 50 cm, and sliced at 2 cm intervals for the first 10 cm, then 5 cm intervals until the end. The preparation and analysis of samples followed standard procedures as described by Lubis (2006).

The constant rate of supply (CRS) technique developed by Appleby and Oldfield (1978) that has been used elsewhere (Lubis, 2006; Sanchez-Cabeza and Ruiz-Fernández, 2012; Sasmito et al., 2020a) was adopted. It is assumed that ^{210}Pb can be formed in wetlands from in situ decay of ^{226}Ra (supported ^{210}Pb); ^{210}Pb can also be indirectly deposited in wetlands via the water column, from the ex situ decay of ^{222}Rn in the atmosphere (unsupported ^{210}Pb) (Villa and Mitsch, 2014). We consider supported ^{210}Pb activity to be similar to total ^{210}Pb found in the lower segments of the soil profile (lowest value in the soil profile); while unsupported ^{210}Pb activity was estimated by subtracting the total ^{210}Pb from supported ^{210}Pb activity, for each core in the soil profile (Craft and Richardson, 1998; Cossa et al., 2014; Villa and Mitsch, 2014; Sasmito et al., 2020a).

The dating analysis followed the below equation (Appleby and Oldfield, 1978),

$$A = Ao e^{-kt} \tag{1}$$

where A is unsupported ^{210}Pb activity below the individual segment being dated (Bq kg^{-1}), Ao is total unsupported ^{210}Pb activity in the soil column (Bq kg^{-1}), k is the ^{210}Pb day constant (0.0311 yr^{-1}), and t is the age of sediment (yr) in each segment. The age of sediment was derived by:

$$t = \frac{1}{k} ln \frac{Ao}{A} \tag{2}$$

Sediment mass accumulation (g m^{-2}yr^{-1}) was calculated by dividing interval mass (g m^{-2}) by the age of sediment in each interval. The accretion rate (mm yr^{-1}) was obtained by dividing mass accumulation (g m^{-2} yr^{-1}) by the bulk density (g cm^{-3}) of the sediment intervals (Smoak et al., 2012; Breithaupt et al., 2014). Historical carbon burial rates were calculated by multiplying the accretion rate (mm yr^{-1}), bulk density (g cm^{-3}), and carbon concentration (g C g-soil^{-1}) (Marchio et al., 2016).

2.6. Statistical analysis

Analysis of Variance (ANOVA) was used to determine differences in each carbon pool as well as surface elevation changes between sites. A post hoc pairwise comparison was also used to show the difference in comparison with one other. Due to certain limitations, there were not enough replications across all parameters to run a two-way ANOVA (e.g. SAR and C Burial rates in hydrogeomorphic settings). Prior to ANOVA, the distribution of data was tested using the Shapiro-Wilk method. When data did not follow a normal distribution pattern, the Kruskal-Wallis non-parametric significance test was performed to determine differences between sites. A p-value of 0.05 was used to determine statistical significance. All analyses were run using IBM-SPSS statistical software (Version 17.0) and all graphs were prepared using SigmaPlot software (Version 12.0).

3. Results

3.1. Mangrove forest structure

Table 1 shows that average DBH in the conserved Jaring Halus mangrove was the highest out of all the mangroves ($x^2(2) = 77.397$, $p < 0.005$). However, the restored Belawan mangrove had the highest tree density ($F = 4.422$, $p = 0.051$), and the highest basal area (BA), when compared with the other sites ($F = 5.232$, $p = 0.035$). The fact that the degraded Percut mangrove had the lowest values, in terms of all structural properties, indicates that human-induced activities had a direct impact on forest structure.

Species diversity and composition varied significantly across the three mangrove types (Table 1). Jaring Halus conserved mangrove had eight species, dominated by *Xylocarpus granatum* (29.92%), with a Shannon diversity index (H′) of 1.39, a Simpson diversity index (D) of 0.59, and a Pielou evenness index (J′) of 0.77. Belawan restored mangrove had six species, dominated by *Avicennia officinalis* (63%). Compared to Jaring Halus, Belawan had a smaller H′ of 1.08, with a D of 0.41 and a J′ of 0.70. Meanwhile, Percut degraded mangrove had five species and was highly dominated by *Avicennia marina* (58.3%), with a H′ of 0.86, a D of 0.46 and a J' of 0.9.

These indices suggest that Jaring Halus, as a conserved mangrove, was the most diverse. However, after restored for 30 years, Belawan also demonstrated improved species diversity (see Table 1). In contrast, Percut degraded mangrove, dominated by a pioneer species of *A. marina*, had the highest evenness index.

The IVI distribution of species across the three mangroves, which indicates the frequency, density and dominance of species, confirms the above findings. Fig. 2 shows that the IVI distribution in Jaring Halus conserved mangrove resulted in greater species diversity, with *R. apiculata* and *X. granatum* being the most important tree species. While in Belawan's restored mangrove, *A. officinalis* was found to be the most important species. In contrast, in Percut's degraded mangrove the important tree species were *A. marina* and *R. mucronata.*

Table 1
Forest structure in Jaring Halus, Belawan and Percut mangroves.

Plot ID	Relative frequency of species	Average DBH (cm)	Tree density (ha^{-1})	Basal area (m^2 ha^{-1})	Shannon Diversity (H′)	Simpson's Diversity (D)	Pielou's evenness (J′)
Jaring Halus – conserved mangrove							
JH1	Bc (5.6%); Ea (5.7%); Ra (49.1%); Rm (9.4%); Xg (30.2%)	15.2 ± 11.3	725.77 ± 71.09	16.70 ± 15.10	1.20	0.36	0.74
JH2	Aa (5.0%); Ao (24%); Bc (14%); Ea (0.8%); Ra (30.6%); Xg (25.6%)	10.9 ± 6.6	1386.54 ± 84.86	18.15 ± 8.77	1.63	0.90	0.91
JH3	Aa (8.2%); Ao (2.7%); Bc (5.5%); Ea (9.6%); Ra (41.1%); Rm (17.8); Xg (15.1%)	14.6 ± 10.9	855.76 ± 41.91	20.59 ± 12.72	1.79	0.32	0.92
JH4	Aa (31.1%); Ao (3.1%); Bc (4.7%); Ea (3.1%); Ra (3.1%); Rm (1.6%); Xg (53.1%)	14.8 ± 9.3	855.76 ± 43.15	16.71 ± 3.12	1.17	0.49	0.60
JH5	Ao (58.9%); Bc (2.2%); Ct (2.2%); Ea (7.8%); Ra (3.3%); Xg (25.6%)	14.1 ± 9.7	1072.40 ± 53.40	22.54 ± 14.66	1.17	0.87	0.66
Mean		*13.92 ± 1.72*	*979.25 ± 58.88*	*18.94 ± 2.56*	*1.39*	*0.59*	*0.77*
Belawan – restored mangrove							
B1	Ao (70%); Bs (14%); Ea (14%)	10.4 ± 6.3	1646.52 ± 16.31	19.62 ± 6.70	0.93	0.45	0.67
B2	Ao (69%); Bs (19%); Ea (8%); Rm (4%)	12.5 ± 8.0	1234.89 ± 107.62	26.20 ± 9.53	0.95	0.45	0.68
B3	Am (13%); Ao (50%); Bs (24%); Ea (9%); Ra (3%); Rm (1%)	11.3 ± 8.2	1115.73 ± 77.64	17.48 ± 9.73	1.36	0.30	0.75
Mean		*11.4 ± 2.26*	*1332.38 ± 67.19*	*21.10 ± 4.55*	*1.08*	*0.41*	*0.70*
Percut – degraded mangrove							
P1	Am (54%); Rm (46%)	11.4 ± 6.7	985.74 ± 73.6	13.56 ± 5.85	0.69	0.50	0.99
P2	Am (77%); Rm (23%)	16 ± 11.8	465.79 ± 45.14	14.42 ± 9.96	0.63	0.55	0.91
P3	Aa (11%); Am (44%); Bs (2%); Ra (14%); Rm (30%)	10.7 ± 8.1	606.61 ± 59.2	8.77 ± 5.17	1.20	0.32	0.79
Mean		*13.04 ± 3.4*	*686.05 ± 59.31*	*12.25 ± 3.04*	*0.86*	*0.46*	*0.90*

Note: Aa (*Avicennia alba*), Ao (*Avicennia officinalis*), Am (*Avicennia marina*), Bc (*Bruguiera cylindrica*), Bs (*Bruguiera sexangula*), Ct (*Ceriops tagal*), Ea (*Excoecaria agallocha*), Ra (*Rhizophora apiculata*), Rm (*Rhizophora mucronata*), Xg (*Xylocarpus granatum*).

3.2. Total ecosystem carbon stocks

A detailed calculation of carbon stocks in all pools and sites is available in Table S2. The highest above ground carbon (AGC) stocks of (mean ± SD) 92.26 ± 22.65 Mg C ha^{-1} were found in Jaring Halus conserved mangrove, while Belawan restored mangrove held 79.40 ± 37.41 Mg C ha^{-1} and Percut degraded mangrove stored 39.89 ± 27.49 Mg C ha^{-1} ($F_{1.3} = 7.069$, $p < 0.017$). The AGC stocks of Belawan mangrove were significantly higher when compared with Percut mangrove ($p = 0.016$), however no significant difference was shown when compared with Jaring Halus mangrove ($p = 0.067$).

Jaring Halus mangrove also had the highest below ground carbon (BGC) stocks at 30.08 ± 6.82 Mg C ha^{-1}, compared to Belawan's stocks of 29.44 ± 13.19 Mg C ha^{-1} and Percut which held 6.67 ± 3.99 Mg C ha^{-1} ($F_{1.3} = 19.202$, $p < 0.001$). The difference in BGC stocks between Belawan restored mangrove and Percut degraded mangrove was significant ($p = 0.016$), however when compared with Jaring Halus conserved mangrove there was not significantly different ($p = 0.800$). These values suggest that conservation maintains above and below-ground biomass in actively growing and functioning mangroves.

Dead wood debris (DWD) carbon stocks across sites did not differ significantly; Belawan's mangrove held the least (4.35 ± 4.74 Mg C ha^{-1}) followed by Percut (7.40 ± 5.58 Mg C ha^{-1}) and Jaring Halus (13.03 ± 8.60 Mg C ha^{-1}) ($F_{1.3} = 2.196$, $p = 0.174$). The DWD stocks value also was not significantly different between Belawan restored mangrove and Percut degraded mangrove ($p = 0.789$) or Jaring Halus conserved mangrove ($p = 0.285$). While it seems quite dynamic, DWD is the smallest among other carbon pools.

The least soil organic carbon (SOC) stocks, however, were found in the conserved mangrove of Jaring Halus (127.49 ± 33.21 Mg C ha^{-1}), where stocks were even lower than in the degraded mangrove of Percut (139.17 ± 25.44 Mg C ha^{-1}), and the restored mangrove of Belawan which held the most across all mangrove sites (236.26 ± 20.33 Mg C ha^{-1}) (F1.3 = 10.93, $p < 0.01$). The SOC stock levels were not significantly different when comparing between Belawan restored mangrove and Percut degraded mangrove ($p = 0.139$), or Jaring Halus conserved mangrove ($p = 0.150$). Environmental settings or types of mangrove may have affected the soil carbon dynamic.

Total ecosystem carbon stocks (TECS) across the three different mangroves are shown in Fig. 3. The Jaring Halus conserved mangrove had TECS of 262.29 ± 69.71 Mg C ha^{-1}, less than seen in the Belawan restored mangrove which had 349.44 ± 75.68 Mg C ha^{-1}, but more than in Percut's degraded mangrove which held 193.12 ± 62.50 Mg C ha^{-1} ($F_{1.3} = 2.065$, $p = 0.189$). While restoration improved TECS considerably, mangrove conservation was also seen to maintain species biodiversity. However, 30 years of restoration only slightly improved the diversity indices, as shown in Table 1. The degraded mangrove in Percut had lost considerable AGC, but still maintained a substantial amount of its BGC.

3.3. Contemporary sedimentation

The rSET-MH measurements revealed both positive and negative surface elevation change (SEC); subtle sub-surface changes were noted through rSET measurements, while MH measurements were always positive (Table 2). The averaged interior SEC in Percut degraded mangrove was the highest at 41.7 ± 24.0 mm yr^{-1}, followed by Belawan restored mangrove (20.7 ± 14.6 mm yr^{-1}) and Jaring Halus conserved mangrove (12.2 ± 3.9 mm yr^{-1}) ($F_{1.3} = 1.241$, $p = 0.324$). There were no significant differences of surface accretion rate (SAR) and shallow subsidence (SS) across the three mangroves (Jaring Halus, Belawan, Percut) (($F_{1.3} = 1.224$, $p = 0.350$), ($F_{1.3} = 1.117$, $p = 0.379$)). Despite Jaring Halus being an island, there were no significant differences in SEC, SAR and SS compared to the mainland mangroves, Belawan ($p = 0.516$, $p = 0.0.481$, $p = 0.633$ respectively) and Percut ($p = 0.945$, $p = 0.399$, $p = 0.413$ respectively). However, a significant difference was

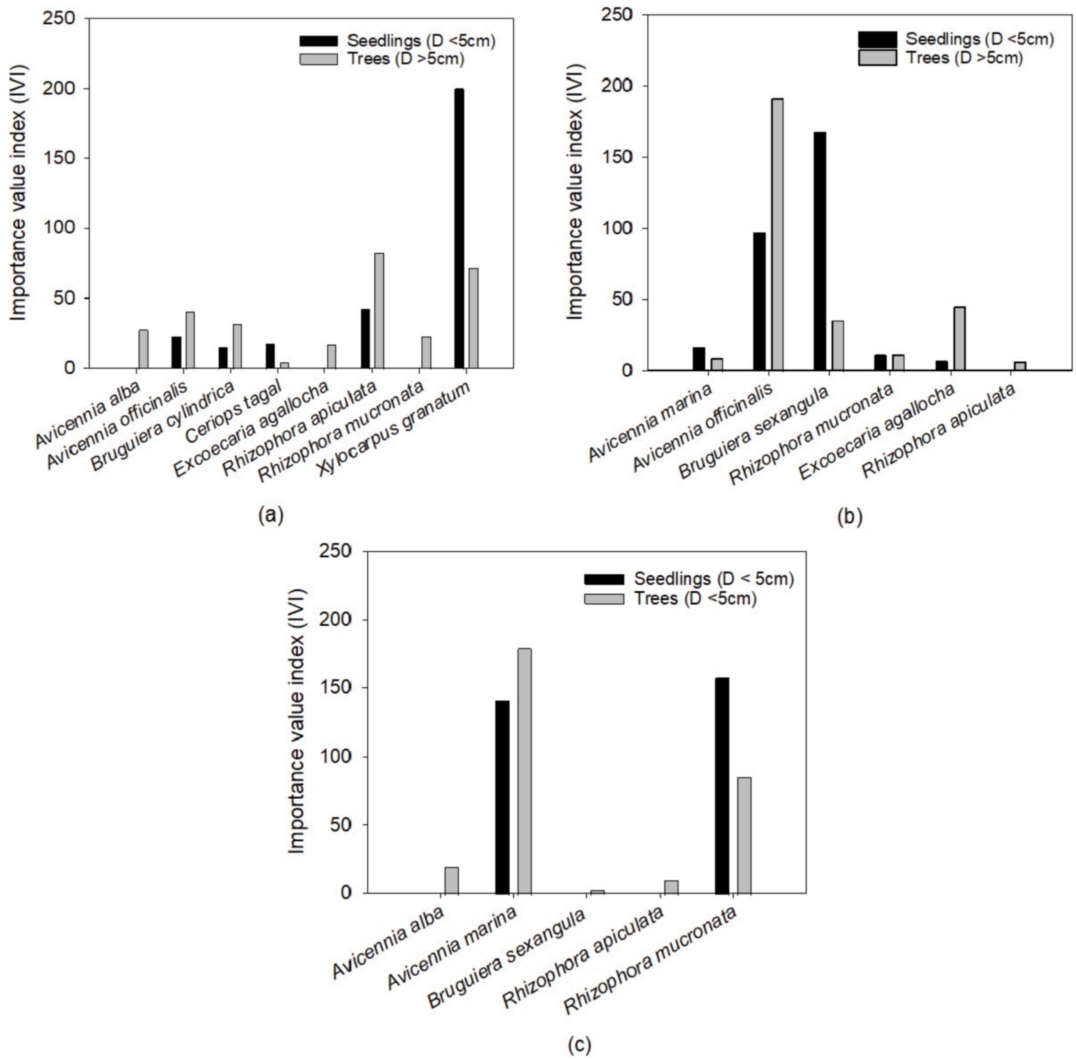

Fig. 2. The importance value index (IVI) of mangrove species in (a) Jaring Halus (conserved mangrove), (b) Belawan (restored mangrove), and (c) Percut (degraded mangrove).

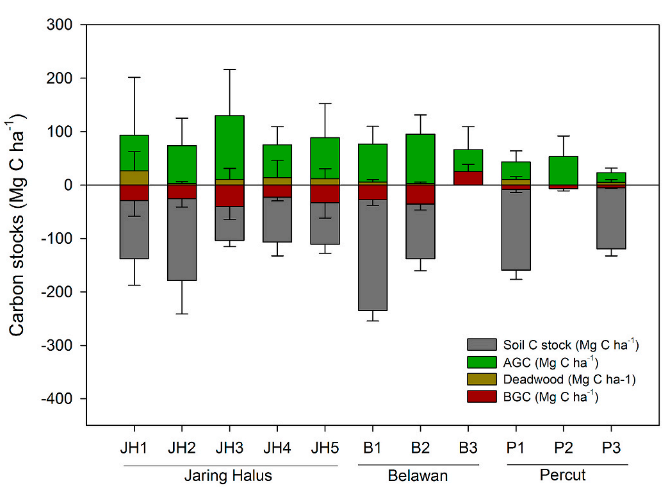

Fig. 3. A comparison of total carbon stocks across Jaring Halus (conserved mangrove), Belawan (restored mangrove) and Percut (degraded mangrove) sites, including above and belowground carbon, deadwood and soil carbon stock. Negative sign on Y axis indicates carbon stocks below the surface, which consist of root (brown) and soil (gray).

found in SAR across the different hydrogeomorphic settings (mudflat, fringe, interior) ($F_{1,3} = 15.266$, $p = 0.003$). Despite it not being possible to compare a complete set of data across all hydrogeomorphic settings, we did find that fringe mangroves tended to have higher SAR than interior mangroves.

3.4. Historical sedimentation

Detailed results from the radionuclide analysis of soil cores taken from Jaring Halus conserved and Percut degraded mangroves are presented in Table S3. These results show that a difference in hydrogeomorphic setting leads to a fluctuation in ^{210}Pb activity. Mudflats show considerable fluctuations across a short range of values; fringe mangroves fluctuate less in their lower layers, but have wider ranging values, while interior mangroves show a clear pattern of values decreasing with depth. The later confirms the simulated pattern of decreasing activities (Arias-Ortiz et al., 2018). Aside from hydrogeomorphic ranges, bioturbation or mixing can also result in these variations. As summarized in Fig. 4, total ^{210}Pb activity in Jaring Halus ranged from 15.85 to 86.85 Bq kg^{-1} across mudflat, fringe and interior mangroves, while in Percut, this ranged from 26.21 to 78.74 Bq kg^{-1}.

Table 3 summarizes the accumulation rates in Jaring Halus conserved mangrove, across its mudflat, fringe and interior locations, at 0.19 ± 0.15 g cm^{-2}yr^{-1}; 0.25 ± 0.05 g cm^{-2}yr^{-1}; and 0.47 ± 0.51 g

Table 2
The (mean ± SD) annual rates of contemporary surface elevation change (SEC), surface accretion rate (SAR), shallow subsidence (SS), and carbon burial rates across study sites, calculated using the rSET-MH technique.

Site	Hydrogeomorphic setting	rSET-MH station	Measurement duration (years)	Surface elevation change (SEC) (mm yr^{-1}) *	Surface accretion rate (SAR) (mm yr^{-1})	Shallow subsidence (SS)* (mm yr^{-1})	Carbon burial rates (g C m^2 yr^{-1})
Jaring Halus	Mudflat	M-1	1.8	9.8 ± 5.2	99.5 ± 13.5	89.7 ± 13.5	1595.87 ± 225.16
	Fringe	F-1	1.8	28.8 ± 4.8	42.8 ± 5.1	14.0 ± 5.1	540.38 ± 64.29
	Fringe	F-2	1.8	5.9 ± 4.1			
	Interior	I-1	1.8	20.4 ± 5.5	15.1 ± 10.9	−5.3 ± 10.9	168.55 ± 38.56
	Interior	I-2	1.8	15.8 ± 3.1			
	Interior	I-3	1.8	0.5 ± 3.1	25.8 ± 5.7	25.3 ± 5.7	466.99 ± 32.63
Mean fringe				*17.4 ± 4.5*	*42.8 ± 5.1*		*168.55 ± 121.93*
Mean interior				*12.2 ± 3.9*	*20.4 ± 0.83*	*10.0 ± 8.3*	*317.77 ± 188.48*
Belawan	Fringe	F-1	3.5	−3.0 ± 5.1			
	Fringe	F-2	3.5	−22.6 ± 3.5			
	Interior	I-1	3.5	36.2 ± 20.7	10.3 ± 9.7	−25.9 ± 9.7	238.62 ± 225.02
	Interior	I-2	3.5	5.2 ± 8.5	40.2 ± 10.1	34.9 ± 10.1	929.38 ± 234.63
Mean fringe				*−18.2 ± 4.3*			
Mean interior				*20.7 ± 14.6*	*25.3 ± 9.9*	*4.5 ± 43*	*584.00 ± 488.44*
Percut	Mudflat	M-1	3.5	45.7 ± 8.6			
	Fringe	F-1	3.5	−15.1 ± 23.8	16.5 ± 0.9	31.6 ± 0.9	153.55 ± 8.45
	Fringe	F-2	3.5	26.3 ± 15.4	29.5 ± 7.5	3.2 ± 7.5	274.26 ± 69.74
	Interior	I-1	3.5	13.2 ± 4.2	0.00	−13.2	
	Interior	I-2	3.5	70.2 ± 43.8	19.2 ± 7.9	−51.0 ± 7.9	289.63 ± 118.93
Mean fringe				*5.6 ± 19.6*	*23.0 ± 4.2*	*17.4 ± 20*	*234.02 ± 91.38*
Mean interior				*41.7 ± 24.0*	*9.6 ± 3.9*	*−32.1 ± 26*	*289.63 ± 118.93*

* Negative values represent subsurface expansion, whereas positive values represent subsidence.

$cm^{-2}yr^{-1}$ respectively ($p = 0.197$). A similar pattern is seen in Percut degraded mangrove, where accumulation rates are 0.24 ± 0.08 g $cm^{-2}yr^{-1}$; 0.39 ± 0.41 g $cm^{-2}yr^{-1}$; and 0.21 ± 0.08 g $cm^{-2}yr^{-1}$ across the same settings ($p = 0.206$). The average sediment accumulation rate in Jaring Halus was 0.32 ± 0.55 g $cm^{-2}yr^{-1}$, while in Percut this was 0.29 ± 0.27 g $cm^{-2}yr^{-1}$ ($p = 0.810$).

In Percut degraded mangrove, sediment accretion in the fringe mangrove averaged 8.56 ± 8.62 mm yr^{-1}, with less sediment accretion seen in the mudflat (4.78 ± 1.67 mm yr^{-1}) and interior (4.28 ± 2.26 mm yr^{-1}) ($p = 0.35$). Meanwhile, Jaring Halus conserved mangrove saw sediment accretion of 9.80 ± 6.93 mm yr^{-1} in its interior mangrove, followed by the fringe (7.60 ± 4.12 mm yr^{-1}) and mudflat sites (4.75 ± 4.73 mm yr^{-1}) ($p = 0.241$). Average accretion rates were 7.73 ± 10.09 mm yr^{-1} and 5.87 ± 5.86 mm yr^{-1} for Jaring Halus and Percut respectively ($p = 0.693$). Although we cannot compare with the accretion rates obtained using a contemporary approach, it seems that recent disturbance has been more severe and has caused higher sediment loss than that of 50–60 years ago, as the age of sediment indicates.

3.5. Contemporary and historical carbon burial rates

Table 2 shows a summary of the overall average of contemporary carbon burial rates. The Jaring Halus conserved mangrove has an average value of 567.72 ± 530.97 g C m^2 yr^{-1}, while the average contemporary carbon burial rate at Belawan restored mangrove was 584.00 ± 488.44 g C m^2 yr^{-1}, and Percut degraded mangrove was 247.93 ± 79.62 g C m^2 yr^{-1} ($F_{1.3} = 0.469$, $p = 0.640$). There was no significant difference in carbon burial values between Jaring Halus and Belawan ($p = 0.678$) or Percut mangrove ($p = 0.359$). This is also in line with the insignificant difference in carbon burial values between Belawan and Percut mangroves ($p = 0.425$). Detailed calculations for these contemporary carbon burial rates, as derived from MH monitoring, are presented in Table S4.

In terms of the historical carbon burial rates, as derived from radionuclide dating, which could only be assessed in two mangroves (see Table 3), results indicate the average historical carbon burial rate in Jaring Halus conserved mangrove was approximately 119.65 ± 190.45 g C m^{-2} yr^{-1} while in Percut degraded mangrove this rate was 62.85 ± 60.15 g C m^{-2} yr^{-1} ($F_{1.2} = 1.867$, $p = 0.176$). The oldest sediment sampled from Percut mangrove was 58 years old (dated back to 1957), which may suggest that the upland disturbance that caused sedimentation was more recent than that found in Jaring Halus, which was dated back to 1912 (see Table S3).

4. Discussion

4.1. Does forest structure matter?

Managing mangrove forests needs comprehensive understanding, as humans intervene through different management regimes. Halophyte coastal vegetation behaves differently to that of neighboring ecosystems in terrestrial settings. Deforested and degraded mangroves are often restored or rehabilitated using standard processes, without sufficient understanding of the ecosystem's structure and functioning (Osland et al., 2012; Adame et al., 2018). These structures and functions have crucial roles in the process of mangrove recovery. This process is influenced by patch size, nutrient availability and species composition (White and Jentsch, 2001). It is noteworthy that water management, including the exchange of fresh and saline water through groundwater seepage, is one of the most important overall factors influencing survival rates (Tack and Polk, 1999; Kairo et al., 2001). Understanding and implementing mangrove species zonation is also essential for determining suitable areas for different species, so as to increase the success rates of mangrove plantation programs (Rabinowitz, 1978).

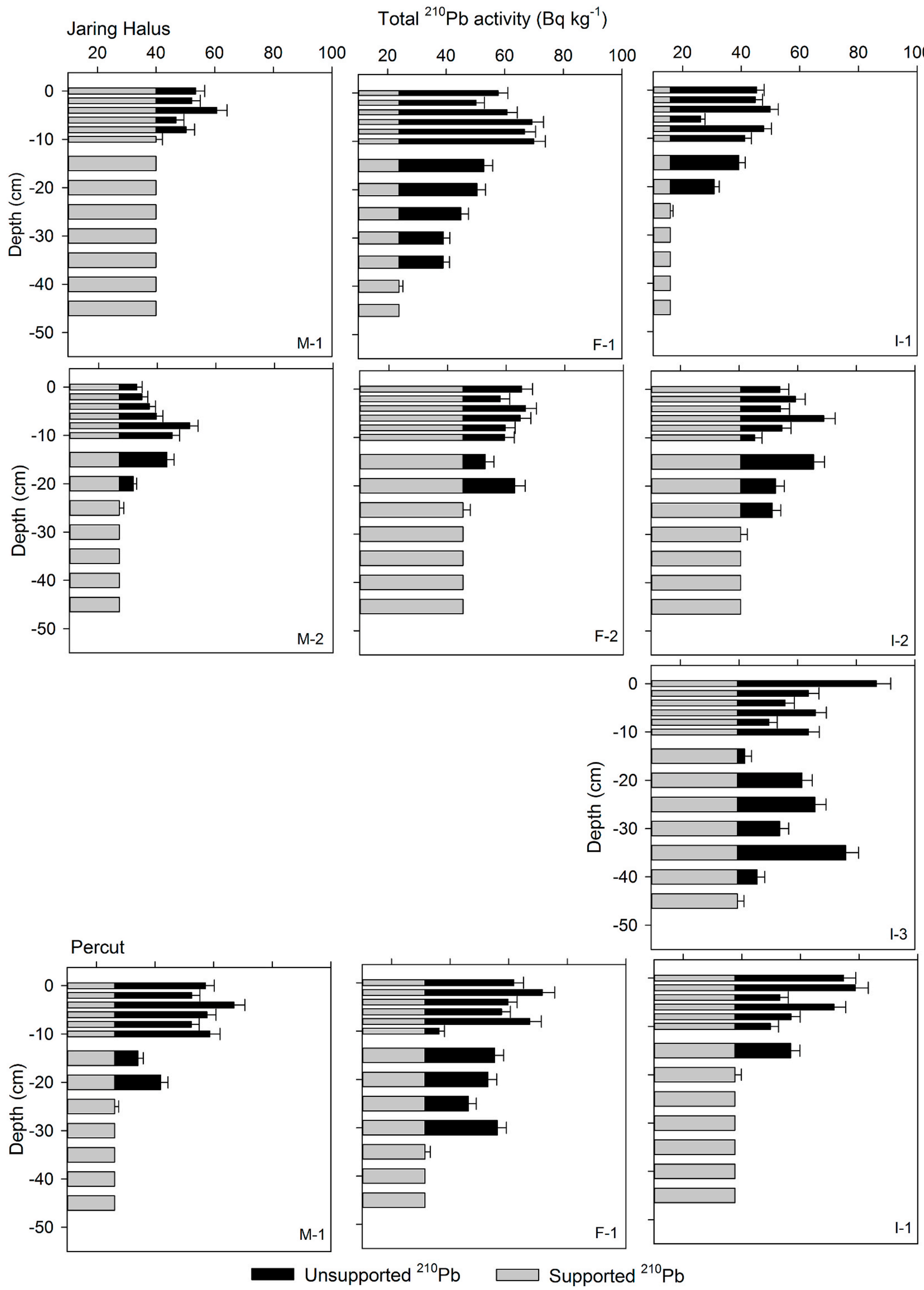

Fig. 4. Total ^{210}Pb activity in sediment across different hydrogeomorphic settings – mudflat (M), fringe (F) and interior (I) – in Jaring Halus conserved mangrove and Percut degraded mangrove. Error bars indicate uncertainties generated through multiple runs of the alpha-spectrometer.

Increasing tree density and basal area should be prioritized in mangrove management, as these two parameters may determine the survival of the stands and protect against adverse environmental impacts (Kamali and Hashim, 2011; Kathiresan et al., 2016). These parameters are also considered as indicators of stand development (population, biomass, and productivity) in response to stress factors (Twilley et al., 1998). In the case of the Belawan restored mangrove, the increased tree species diversity and stand density may have been supported by improvements in salinity (fresh and saline water exchange) and protection from extreme tidal exchange, waves, and wind shear. The Percut mangrove is severely degraded and needs substantial support to maintain its density and species diversity, as reflected by the domination of *Avicennia marina* (Table 1 and Fig. 2). As the development and regeneration pattern of mangrove forests vary from one hydro-geomorphological setting to another, the response to any supporting actions will depend on shoreline topography, sources of sediment, rates of sediment supply, and the rate of sea-level change (Woodroffe et al., 1993).

Table 3
The rates of historical sediment accumulation, vertical accretion and carbon burial in North Sumatran mangroves (Mean ± SD).

Hydro-geomorphic setting	Sediment core	Age (year dated)	Historical accumulation rate (g cm^{-2} yr^{-1})	Historical vertical accretion rate (mm yr-1)	Historical carbon burial rate (g C m^{-2} yr^{-1})
Jaring Halus – conserved mangrove					
Mudflat	M-1	57.6 *(1958)*	0.08 ± 0.04	1.55 ± 1.13	21.78 ± 14.68
Mudflat	M-2	81.9 *(1934)*	0.31 ± 0.26	7.94 ± 7.82	110.87 ± 114.91
Fringe	F-1	74.72 *(1941)*	0.27 ± 0.14	9.79 ± 8.70	90.97 ± 53.53
Fringe	F-2	41.2 *(1975)*	0.23 ± 0.07	5.42 ± 2.87	43.26 ± 16.35
Interior	I-1	65.2 *(1951)*	0.23 ± 0.17	5.56 ± 4.26	72.98 ± 66.71
Interior	I-2	61.9 *(1954)*	0.40 ± 0.30	9.01 ± 7.20	107.12 ± 79.56
Interior	I-3	104.2 *(1912)*	0.78 ± 1.11	14.85 ± 17.46	274.31 ± 371.00
Mean mudflat			0.19 ± 0.17	4.75 ± 4.52	66.33 ± 63.00
Mean fringe			0.25 ± 0.03	7.60 ± 3.09	67.11 ± 33.74
Mean interior			0.47 ± 0.28	9.80 ± 4.70	151.47 ± 107.74
Percut – degraded mangrove					
Mudflat	M-1	54.4 *(1962)*	0.24 ± 0.08	4.78 ± 1.67	45.06 ± 16.10
Fringe	F-1	58.6 *(1957)*	0.39 ± 0.41	8.56 ± 8.62	81.71 ± 90.54
Interior	I-1	42 *(1974)*	0.21 ± 0.08	4.28 ± 2.26	55.33 ± 24.84

Our hypothesis is supported by the facts that Jaring Halus conserved mangrove had higher values of AGC, BGC and DWD. Even though the value is small, conserved mangrove continued to see deadwood debris being deposited from elsewhere and trapped inside the mangrove. In addition, Belawan mangrove also proved that restoration has improved both above and belowground carbon stocks. This implies that while conservation is always the best management strategy, restoration may be an effective way to improve carbon stocks and tree diversity, when compared to continual degradation. The results were in line with studies by Sasmito et al., (2020b) in West Papua, Indonesia, and by Sharma et al., (2020) in Cambodia, which found that mangroves left to regenerate or those restored for more than 25 years could reach the same levels of biomass carbon as undisturbed mangrove. One important result is the anomaly of SOC stocks, which saw the conserved mangrove of Jaring Halus storing less carbon ($F_{1,3} = 10.93$, $p < 0.01$) than two other sites. This may be related to the island location of Jaring Halus mangrove; both Belawan and Percut mangroves are well connected with the mainland, and carbon cycling may take place through different and more efficient pathways.

Improving mangrove forest structure could be a prerequisite of future restoration programs, with the specific hydrogeomorphic setting determining the site requirements, i.e. the particular species to be (re) introduced (Lewis, 2005; Sasmito et al., 2019). Impacts of the first 30 years of rehabilitation efforts have been revealed, as summarized in Table 1. Similar results were seen in a study assessing the effectiveness of restoration on TECS in Cambodia, over 25 years (Sharma et al., 2020).

4.2. The effectiveness of different management regimes in terms of carbon burial

In addition to structural improvements in forests, restoration programs could also consider sediment accretion and stability. Sedimentation is a good indicator of the potential services that the mangrove ecosystems could provide; in particular vertical expansion means the initiation of coastal fortification, to adjust to rising sea levels (Krauss et al., 2013) as well as mangrove ecosystems acting as effective carbon sinks in coastal areas (Smoak et al., 2012). Depending on the source and origin of the carbon, sedimentation can also be associated with carbon burial, as in time, sedimentation can result in large carbon accumulation or dilute carbon stocks (MacKenzie et al., 2016; Soper et al., 2019). These are slow processes to evaluate stability, but historically traceable (as indicated by radionuclide dating) and contemporary elevation change (indicated by rSET-MH), and give an understanding of the dynamics and processes (Table 3 and Fig. 4). Sedimentation as a regime is related to a complex geological process, influenced not only by aboveground processes by also by groundwater dynamics (Semeniuk, 1994). Belowground soil and hydrology dynamics drive wave base levels and frequency of inundation according to sea level. In turn, these aspects fundamentally change the soil and salinity regime, which affects the process of shaping the shore profile, as well as sediment deposition in mangroves (Semeniuk, 1994).

From the two contrasting settings of the conserved mangrove in Jaring Halus and the degraded mangrove in Percut, it can be seen that carbon burial rates in the interior of the conserved mangrove were higher than carbon burial rates in the degraded mangroves, based on both contemporary (Table 2) and historical (Table 3) results. This indicates that the conserved mangrove of Jaring Halus, with its higher AGC and BGC, was able to bury carbon much faster than the degraded mangrove of Percut. The contribution of decomposed deadwood, twigs, leaves and roots to buried carbon (Krauss et al., 2013) was significant. However, fringe mangroves have accumulated more carbon-containing sediment than interior mangroves and mudflats in the Percut degraded mangrove.

While the rSET-MH indicates uncertainties associated with not capturing the temporal dynamics of the coastal environment, the ^{210}Pb technique had uncertainties resulting from model limitations, soil section scales, and dating technique bias (Breithaupt et al., 2020). In this study, the rate of contemporary carbon burial was approximately twice that of historical carbon burial rates. Although the size is not comparable, as the methods used were different, the trend was similar. This trend is quite useful, therefore, to understand future sedimentation rates based on past and present-day measurements.

Based on Tables 2 and 3, higher carbon burial values were always found in Jaring Halus conserved mangrove, as opposed to Percut degraded mangroves, even though there were no significant differences ($F_{1,3} = 0.469$, $p = 0.640$). Globally, however, sediment accretion and carbon burial vary significantly according to location, hydrogeomorphic and environmental setting. As shown in Fig. 5, sampling sites with high sediment accretion do not necessarily have high carbon burial, and vice versa. This is because not all sediments that settle in mangroves contain high levels of organic matter. On the other hand, sediment that contains certain minerals can cause dilution, which reduces existing carbon (Kusumaningtyas et al., 2019). Local factors, such as tides, waves, geomorphological and environmental settings, play a role in controlling the distribution and depositing of sediment that incorporates carbon in intertidal mangroves (Krauss et al., 2013; Woodroffe et al., 2016; Kusumaningtyas et al., 2019). Such factors determine the accommodation space, where sediment accretes over time, which is usually greatest in seaward edges and smallest in landward edges, due to the high spring tides that reach both these areas (Woodroffe et al., 2016).

Globally, carbon burial rates average at 163.3 (+40; −31) g C m^{-2}

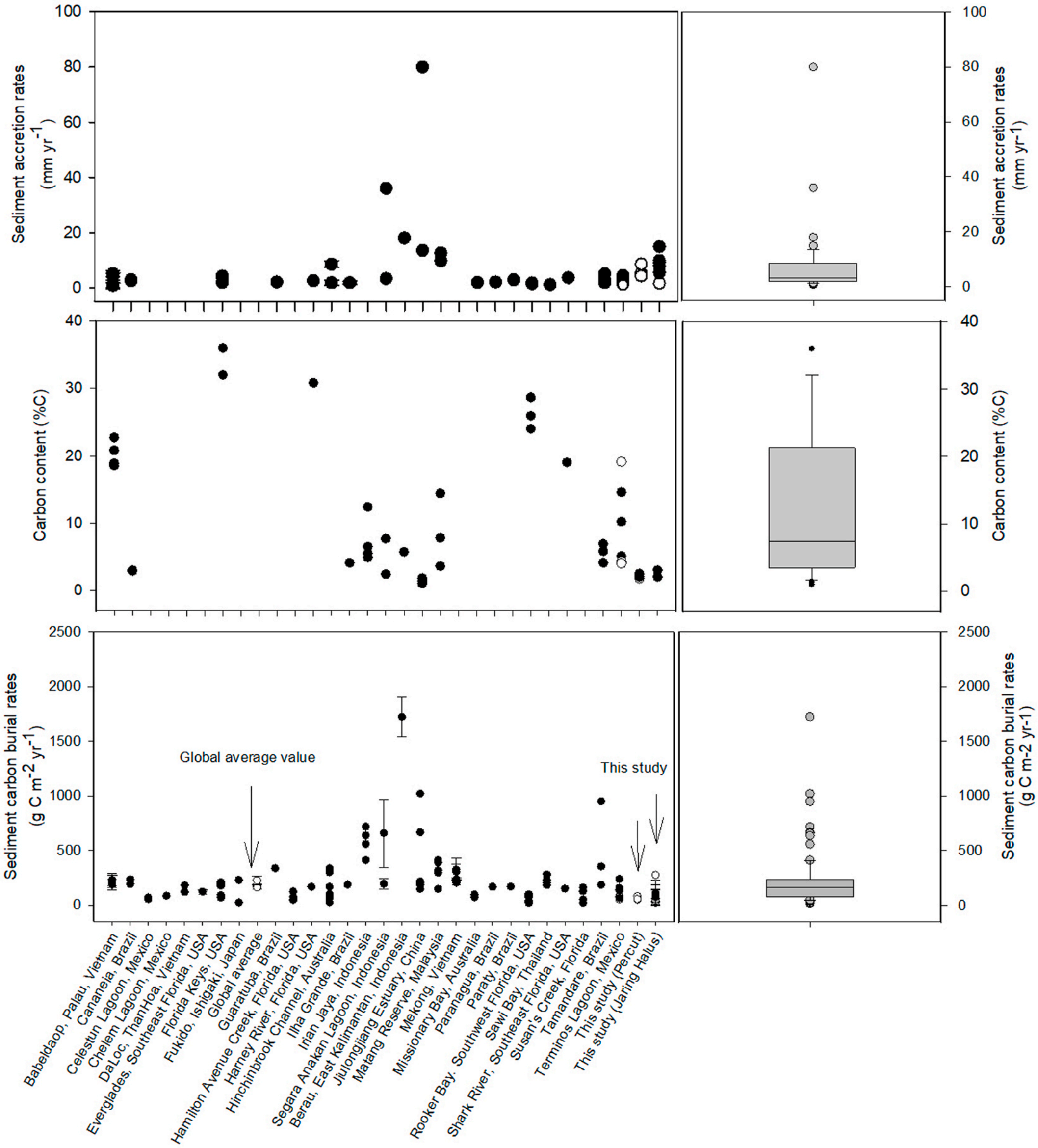

Fig. 5. Historical sediment accretion, sediment carbon content and soil carbon accumulation rates in Jaring Halus and Percut (open circle), compared with values in other sites (solid circles) generated using radio-nuclide techniques by Lynch et al. (1989), Cahoon and Lynch (1997), Callaway et al. (1997), Alongi et al. (1999), Alongi et al. (2001), Alongi et al. (2004), Brunskill et al. (2004), Gonneea et al. (2004), Alongi et al. (2005), Tateda et al. (2005), Sanders et al. (2008), Sanders et al. (2010a), Sanders et al. (2010b), Sanders et al. (2010c), Smoak et al. (2012), MacKenzie et al. (2016) and Marchio et al. (2016), and Kusumaningtyas et al. (2019). Modified from Breithaupt et al. (2012) and Marchio et al. (2016).

yr^{-1} (Breithaupt et al., 2012). Carbon burial rates from this study were in the 10th percentile when compared with global average values. This outcome is unsurprising, given that the study sites may have been affected by human-induced activities. As a comparison, the average soil carbon content of North Sumatran mangroves ranges between 1.5% and 4.4%, far lower than the soil carbon content found in mangroves located in Sumatra (9.4%), Kalimantan (9.7%) and Sulawesi (15.6%), and even lower than in degraded mangroves in Java, which measure a soil carbon content of 5.6% (Murdiyarso et al., 2015).

Aside from the sources of sediment, biophysical perturbation effects (e.g. erosion, deposition, pedoturbation) also result in less stable sediments, especially in low soil bulk density areas (Sasmito et al., 2019). In general, higher sedimentation rates were recorded in fringe mangrove areas than in interior or mudflat areas. This is most likely due to plant structures reducing water velocity, and stagnant zones being created between roots and trunks (MacKenzie et al., 2016). Sediment stabilization, supported by vegetation in fringe and interior, reduced the fluctuating pattern of ^{210}Pb activity (See Fig. 4) (Sasmito et al., 2019).

Carbon burial rates that have large ranges and deviation (in Fig. 5) reflect the complexity of this process. As well as environmental settings, changes in relative sea level rise (SLR) also become a driver of increased carbon burial rates (Rogers et al., 2019; Watanabe et al., 2019). On the other hand, biological factors (roots growth, decomposition rates, benthic mat formation), environmental factors (salinity, nutrients, flooding, soil texture), and physical factors and feedbacks (rainfall variability, response to atmospheric CO_2, sea level rise) also affect the sedimentation and/or carbon burial processes (Krauss et al., 2013). Further study requires radionuclide isotopes to better understand the dynamics of sedimentation and carbon burial processes.

4.3. Blue carbon potential

Understanding of blue carbon ecosystems, including mangrove, seagrass and salt marshes, is increasing as these systems have shown themselves to be nature-based solutions to mitigate climate change (Murdiyarso et al., 2015). This understanding must be deepened, however, in the context of avoiding the deforestation and degradation of mangroves. Given the key role this ecosystem plays as a large carbon storage solution, it is critical to avoid releasing the carbon stored in mangrove soil and biomass.

The Indonesian government has proposed a 29% greenhouse gas emission reduction by 2030, with a major contribution (60%) expected from the forestry sector (Republic of Indonesia, 2016). Enhancing the sink capacity of an ecosystem as promising as mangroves is key to achieving this emission reduction target. Restoring degraded mangroves and abandoned fishponds provides multiple benefits, including livelihood options for the community. In addition, conserving the remaining mangrove areas could avoid further deforestation and ensure the sustainability of these unique ecosystems. Given the variations in land-use intensity across coastal landscapes, options for best practice management should be aligned with local agendas.

Although the carbon burial rates in this study are lower than the global average, our study suggests that a substantial amount of carbon could be sequestered annually by mangrove ecosystems. The island of Jaring Halus with its conserved mangrove area showed an ability to store carbon in sedimentation at rates of 66.33 ± 70.87 g C $m^{-2}yr^{-1}$ in its mudflat, 67.11 ± 26.29 g C $m^{-2}yr^{-1}$ in its fringe and 151.47 ± 172.09 g C $m^{-2}yr^{-1}$ in its interior mangrove ecosystems. These numbers are equal to the removal of 2.4–5.5 Mg CO_2e ha^{-1} y^{-1}, which is a meaningful contribution for such a small island. Further, restoring degraded mangrove in the area of Belawan could potentially reduce as much as 3.5 Mg CO_2e ha^{-1} y^{-1} in emissions. As such, these blue carbon areas offer potential funding streams, through either voluntary or mandatory market mechanisms.

This data supports the generation of new emission factors to be contributed to the Intergovernmental Panel on Climate Change (IPCC) database. Diverse land-use management, including conservation and the restoration/rehabilitation of intact or degraded mangroves, requires specific emission factors. Adopting the IPCC Guidelines is encouraged, to allow the use of country-specific and/or site-specific emission factors from the land-use sector (IPCC, 2014).

5. Conclusions

Thirty years of mangrove restoration in North Sumatra has had positive effects in terms of improving forest structure and species composition. Species diversity is now closer to that seen in conserved mangroves with higher tree densities, and carbon stocks have been substantially improved across all pools, when compared with unrestored degraded mangroves.

Improvements in vegetation structure and composition have also enhanced the capacity of restored and conserved mangroves to accrete sediment, and therefore build land to cope with rising sea levels. Since sediment also contains a reasonable amount of carbon, accretion likewise helps to improve long-term soil carbon storage.

While the objective of restoration is not just to improve carbon stocks so as to meet climate change objectives, improvements in biodiversity like this could also lead to improvements in ecosystem functioning. As such, all stakeholders are encouraged to have multiple objectives.

Declaration of competing interest

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

Acknowledgments

This study was part of the Sustainable Wetlands Adaptation and Mitigation Program (SWAMP) funded by the United States Agency for International Development (AID-BFS-IO-17-00005) and Forests, Trees and Agroforestry (FTA), the Research Program of the Consultative Group of International Agricultural Research (CGIAR). The fieldwork, facilitated by the YAGASU Foundation and their field staff, is gratefully acknowledged. The immense support of staff at the Center of Isotopes and Radiation Application, National Nuclear Agency is recognized and thanked. We also thank Sigit Sasmito and Imam Basuki for their helpful comments that have greatly improved this manuscript.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecss.2021.107467.

Author statement

I am, Daniel Murdiyarso, the corresponding author of this manuscript declare that in revising the manuscript I have been consulting and receiving feedback and inputs from all co-authors. On behalf of the co-authors, I would also like to confirm that we have addressed all of the Reviewers' comments.

Sincerely.

Daniel Murdiyarso.

Conflict of interest

Bayu B Hanggara, Daniel Murdiyarso, Yohanes RS Ginting, Yessica L Widha, Grace Y Panjaitan, and Ali A Lubis declare that they have no conflict of interest.

Author contributions

B.B.H led the data analysis, drafted all figures and wrote the manuscript; D.M. conceived the ideas, led the data synthesis/interpretation and wrote the manuscript; Y.S.R.G, Y.L.W. and G.Y.P performed fieldwork and data analysis; A.A.L performed ^{210}Pb laboratory analyses and undertook data interpretation.

References

Adame, M.F., Neil, D., Wright, S.F., Lovelock, C.E., 2010. Sedimentation within and among mangrove forests along a gradient of geomorphological settings. Estuar. Coast Shelf Sci. 86, 21–30. https://doi.org/10.1016/j.ecss.2009.10.013.

Adame, M.F., Brown, C.J., Bejarano, M., Herrera-Silveira, J.A., Ezcurra, P., Kauffman, J. B., Birdsey, R., 2018. The undervalued contribution of mangrove protection in Mexico to carbon emission targets. Conserv. Lett. 11, 1–9. https://doi.org/10.1111/conl.12445.

Alongi, D., Tirendi, F., Dixon, P., Troot, L., Brunskill, G., 1999. Mineralization of organic matter in intertidal sediments of a tropical semi-enclosed delta. Estuar. Coast Shelf Sci. 48, 451–467. https://doi.org/10.1006/ecss.1998.0465.

Alongi, D.M., Wattayakorn, G., Pfitzner, J., Tirendi, F., Zagorskis, I., Brunskill, G.J., Davidson, A., Clough, B.F., 2001. Organic carbon accumulation and metabolic pathways in sediments of mangrove forests in southern Thailand. Mar. Geol. 179, 85–103. https://doi.org/10.1016/S0025-3227(01)00195-5.

Alongi, D., Sasekumar, S., Chong, V., Pfitzner, J., Trott, L., Tirendi, F., Dixon, P., Brunskill, G., 2004. Sediment accumulation and organic material flux in a managed mangrove ecosystem: estimates of land-ocean-atmosphere exchange in peninsular Malaysia. Mar. Geol. 208, 383–402. https://doi.org/10.1016/j.margeo.2004.04.016.

Alongi, D.M., Pfitzner, J., Trott, L.A., Tirendi, F., Dixon, P., Klumpp, D.W., 2005. Rapid sediment accumulation and microbial mineralization in forests of the mangrove Kandelia candel in the Jiulongjiang Estuary, China. Estuar. Coast Shelf Sci. 63, 605–618. https://doi.org/10.1016/j.ecss.2005.01.004.

Appleby, P.G., Oldfield, F., 1978. The calculation of lead-210 dates assuming a constant rate of supply of unsupported ^{210}Pb to the sediment. Catena 5, 1–8. https://doi.org/10.1016/S0341-8162(78)80002-2.

Arias-Ortiz, A., Masqué, P., Garcia-Orellana, J., Serrano, O., Mazarrasa, I., Marbà, N., Lovelock, C.E., Lavery, P.S., Duarte, C.M., 2018. Reviews and syntheses: 210Pb-derived sediment and carbon accumulation rates in vegetated coastal ecosystems – setting the record straight. Biogeosciences 15, 6791–6818. https://bg.copernicus.org/articles/15/6791/2018/.

Bomer, E.J., Wilson, C.A., Hale, R.P., Hossain, A.N.M., Rahman, F.M.A., 2020. Surface elevation and sedimentation dynamics in the Ganges-Brahmaputra tidal delta plain, Bangladesh: evidence for mangrove adaptation to human-induced tidal amplification. Catena 187. https://doi.org/10.1016/j.catena.2019.104312, 104312.

Bouillon, S., Borges, A.V., Castañeda-Moya, E., Diele, K., Dittmar, T., Duke, N.C., Kristensen, E., Lee, S.Y., Marchand, C., Middelburg, J.J., Rivera-Monroy, V.H., Smith, T.J., Twilley, R.R., 2008. Mangrove production and carbon sinks: a revision of global budget estimates. Global Biogeochem. Cycles 22, 1–12. https://doi.org/10.1029/2007GB003052.

BPS North Sumatra, 2015. Sumatera Utara Dalam Angka 2015 (North Sumatera in Figures 2015). Badan Pusat Statistik Provinsi Sumatera Utara, Medan, Indonesia.

Breithaupt, J.L., Smoak, J.M., Smith, T.J., Sanders, C.J., Hoare, A., 2012. Organic carbon burial rates in mangrove sediments: strengthening the global budget. Global Biogeochem. Cycles 26, 1–11. https://doi.org/10.1029/2012GB004375.

Breithaupt, J.L., Smoak, J.M., Smith, T.J., Sanders, C.J., 2014. Temporal variability of carbon and nutrient burial, sediment accretion, and mass accumulation over the past century in a carbonate platform mangrove forest of the Florida Everglades. J. Geophys. Res. Biogeosci. 1–17. https://doi.org/10.1002/2014JG002715.

Breithaupt, J.L., Smoak, J.M., Bianchi, T.S., Vaughn, D.R., Sanders, C.J., Radabaugh, K. R., Osland, M.J., Feher, L.C., Lynch, J.C., Cahoon, D.R., Anderson, G.H., Whelan, K. R.T., Rosenheim, B.E., Moyer, R.P., Chambers, L.G., 2020. Increasing rates of carbon burial in Southwest Florida coastal wetlands. J. Geophys. Res. Biogeosci. 125, 1–25. https://doi.org/10.1029/2019JG005349.

Brunskill, G.J., Zagorskis, I., Pfitzner, J., Ellison, J., 2004. Sediment and trace element depositional history from the Ajkwa river estuarine mangroves of Irian Jaya (West Papua), Indonesia. Continent. Shelf Res. 24, 2535–2551. https://doi.org/10.1016/j.csr.2004.07.024.

Cahoon, D.R., Reed, D.J., Day, J.W., 1995. Estimating shallow subsidence in microtidal salt marshes of the southeastern United States: Kaye and Barghoorn revisited. Mar. Geol. 128, 1–9. https://doi.org/10.1016/0025-3227(95)00087-F.

Cahoon, D., Lynch, J., 1997. Vertical accretion and shallow subsidence in a mangrove forest of southwestern Florida. U.S.A. Mangroves Salt Marshes 1, 173–186. https://doi.org/10.1023/A:1009904816246.

Cahoon, D.R., Lynch, J.C., Perez, B.C., Segura, B., Holland, R.D., Stelly, C., Stephenson, G., Hensel, P., 2002. High-precision measurements of wetland sediment elevation: II. The rod surface elevation table. J. Sediment. Res. 72, 734–739. https://doi.org/10.1306/020702720734.

Callaway, J.C., Delaune, R.D., Patrick, W.H., 1997. Sediment accretion rates from four coastal wetlands along the Gulf of Mexico. J. Coast Res. 13, 181–191. https://doi.org/10.2307/4298603.

Cintron, G., Schaeffer-Novelli, Y., 1984. Methods of studying mangrove structure. In: Snedaker, S.C., Snedaker, J.C. (Eds.), The Mangrove Ecosystem: Research Methods. UNESCO, Paris, pp. 91–113.

Cossa, D., Buscail, R., Puig, P., Chiffoleau, J.F., Radakovitch, O., Jeanty, G., Heussner, S., 2014. Origin and accumulation of trace elements in sediments of the northwestern Mediterranean margin. Chem. Geol. 380, 61–73. https://doi.org/10.1016/j.chemgeo.2014.04.015.

Craft, C.B., Richardson, C.J., 1998. Recent and long-term organic soil accretion and nutrient accumulation in the Everglades. Soil Sci. Soc. Am. J. 62, 834–843.

Datta, D., Deb, S., 2017. Forest structure and soil properties of mangrove ecosystems under different management scenarios: experiences from the intensely humanized landscape of Indian Sunderbans. Ocean Coast Manag. 140, 22–33. https://doi.org/10.1016/j.ocecoaman.2017.02.022.

FAO, 2007. The World's Mangroves 1980–2005. FAO Forestry Paper, Rome, Italy, p. 153.

Giri, C., Ochieng, E., Tieszen, L.L., Zhu, Z., Singh, A., Loveland, T., Masek, J., Duke, N., 2011. Status and distribution of mangrove forests of the world using earth observation satellite data. Global Ecol. Biogeogr. 20, 154–159. https://doi.org/10.1111/j.1466-8238.2010.00584.x.

Gonneea, M.E., Paytan, A., 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, 211–227. https://doi.org/10.1016/j.ecss.2004.04.015.

Huxham, M., Kumara, M.P., Jayatissa, L.P., Krauss, K.W., Kairo, J., Langat, J., Mencuccini, M., Skov, M.W., Kirui, B., 2010. Intra- and interspecific facilitation in mangroves may increase resilience to climate change threats. Phil. Trans. Royal. Soc. B. 365, 2127–2135. https://doi:10.1098/rstb.2010.0094.

IPCC, 2014. In: Hiraishi, T., Krug, T., Tanabe, K., Srivastava, N., Baasansuren, J., Fukuda, M., Troxler, T.G. (Eds.), 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands. IPCC, Switzerland.

Kairo, J.G., Dahdouh-Guebas, F., Bosire, J., Koedam, N., 2001. Restoration and management of mangrove systems - a lesson for and from the East African region. South Afr. J. Bot. 383–389. https://doi.org/10.1016/S0254-6299(15)31153-4.

Kamali, B., Hashim, R., 2011. Mangrove restoration without planting. Ecol. Eng. 37, 387–391. https://doi.org/10.1016/j.ecoleng.2010.11.025.

Kathiresan, K., Saravanakumar, K., Anburaj, R., 2016. A simple method for assessing mangrove forest based on young plants and sesarmid crab holes. Reg. Stud. Mar. Sci. 7, 204–210. https://doi.org/10.1016/j.rsma.2016.07.003.

Kauffman, J., Donato, D., 2012. Protocols for the Measurement, Monitoring and Reporting of Structure, Biomass and Carbon Stocks in Mangrove Forests (No. 86). Bogor, Indonesia.

Krauss, K.W., Mckee, K.L., Lovelock, C.E., Cahoon, D.R., Saintilan, N., Reef, R., Chen, L., 2013. How mangrove forests adjust to rising sea level. Tansley Rev. 19–34. https://doi.org/10.1111/nph.12605.

Kusumaningtyas, M., Hutahaean, A., Fischer, H., Pérez-mayo, M., Ransby, D., Jennerjahn, T., 2019. Estuarine, coastal and shelf science variability in the organic carbon stocks, sources, and accumulation rates of Indonesian mangrove ecosystems. Estuar. Coast Shelf Sci. 218, 310–323. https://doi.org/10.1016/j.ecss.2018.12.007.

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

Lovelock, C.E., Duarte, C.M., 2019. Dimensions of blue carbon and emerging perspectives. Biol. Lett. 15, 20180781 https://doi.org/10.1098/rsbl.2018.0781.

Lubis, A.A., 2006. Constant Rate of Supply (CRS) model for determining the sediment accumulation rates in the coastal area using Lead-210. J. Coast Dev. 10, 9–18. https://doi.org/10.4319/lo.2000.45.4.0990.

Lynch, J.C., Meriwether, J.R., McKee, B.A., Vera-Herrera, F., Twilley, R.R., 1989. Recent accretion in mangrove ecosystems based on 137 Cs and ^{210}Pb. Estuaries 12, 284. https://doi.org/10.2307/1351907.

Lynch, J.C., Hensel, P., Cahoon, D.R., 2015. The surface elevation table and marker Horizon technique: a protocol for monitoring wetland elevation dynamics. Natl. Park Serv. https://doi.org/10.13140/RG.2.1.5171.9761.

MacKenzie, R.A., Foulk, P.B., Klump, J.V., Weckerly, K., Purbospito, J., Murdiyarso, D., Donato, D.C., Nam, V.N., 2016. Sedimentation and belowground carbon accumulation rates in mangrove forests that differ in diversity and land use: a tale of two mangroves. Wetl. Ecol. Manag. 24, 245–261. https://doi.org/10.1007/s11273-016-9481-3.

Marchio, D.A., Savarese, M., Bovard, B., Mitsch, W.J., 2016. Carbon sequestration and sedimentation in mangrove swamps influenced by hydrogeomorphic conditions and urbanization in Southwest Florida. Forests 7. https://doi.org/10.3390/f7060116.

Margono, B.A., Potapov, P.V., Turubanova, S., Stolle, F., Hansen, M.C., 2014. Primary forest cover loss in Indonesia over 2000-2012. Nat. Clim. Change. https://doi.org/10.1038/nclimate2277.

Martinuzzi, S., Gould, W.A., Lugo, A.E., Medina, E., 2009. Conversion and recovery of Puerto Rican mangroves: 200 years of change. For. Ecol. Manag. 257, 75–84. https://doi.org/10.1016/j.foreco.2008.08.037.

Millenium Ecosystem Assessment, 2005. Ecosystem and Human Well-Being: Wetlands and Water Synthesis. World Resources Institute, Washington, DC.

Murdiyarso, D., Purbopuspito, J., Kauffman, J.B., Warren, M.W., Sasmito, S.D., Donato, D.C., Manuri, S., Krisnawati, H., Taberima, S., Kurnianto, S., 2015. The potential of Indonesian mangrove forests for global climate change mitigation. Nat. Clim. Change 5, 1089–1092. https://doi.org/10.1038/nclimate2734.

Nellemann, C., Corcoran, E., Duarte, C.M., Valdés, L., De Young, C., Fonseca, L., Grimsditch, G., 2009. Blue carbon - the role of healthy oceans in binding carbon. In: A Rapid Response Assessment. United Nations Environment Programme, GRID-Arendal, p. 80.

Osland, M.J., Spivak, A.C., Nestlerode, J.A., Lessmann, J.M., Almario, A.E., Heitmuller, P.T., Russell, M.J., Krauss, K.W., Alvarez, F., Dantin, D.D., Harvey, J.E., From, A.S., Cormier, N., Stagg, C.L., 2012. Ecosystem development after mangrove wetland creation: plant-soil change across a 20-year chronosequence. Ecosystems 15, 848–866. https://doi.org/10.1007/s10021-012-9551-1.

Pielou, E.C., 1966. The measurement of diversity in different types of biological collections. J. Theor. Biol. 13, 131–144. https://doi.org/10.1016/0022-5193(66)90013-0.

Rabinowitz, D., 1978. Early growth of mangrove seedlings in Panama, and an hypothesis concerning the relationship of dispersal and zonation. J. Biogeogr. 5, 113. https://doi.org/10.2307/3038167.

Republic of Indonesia, 2016. Indonesia: First Nationally Determined Contribution 18.

Richards, D.R., Friess, D.A., 2016. Rates and drivers of mangrove deforestation in Southeast Asia, 2000-2012. Proc. Natl. Acad. Sci. Unit. States Am. 113, 344–349. https://doi.org/10.1073/pnas.1510272113.

Rogers, K., Saintilan, N., Cahoon, D., 2005. Surface elevation dynamics in a regenerating mangrove forest at Homebush Bay, Australia. Wetl. Ecol. Manag. 13, 587–598. https://doi:10.1007/s11273-004-0003-3.

Rogers, K., Kelleway, J.J., Saintilan, N., Megonigal, J.P., Adams, J.B., Holmquist, J.R., Meng, L., Schile-Beers, L., Zawadzki, A., Mazumder, D., Woodroffe, D.D., 2019. Wetland carbon storage controlled by millennial-scale variation in relative sea-level rise. Nature 567 (7746), 91–95. https://doi.org/10.1038/s41586-019-0951-7.

Sanchez-Cabeza, J.A., Ruiz-Fernández, A.C., 2012. ^{210}Pb sediment radiochronology: an integrated formulation and classification of dating models. Geochem. Cosmochim. Acta 82, 183–200. https://doi.org/10.1016/j.gca.2010.12.024.

Sanders, C.J., Smoak, J.M., Naidu, A.S., Patchineelam, S.R., 2008. Recent sediment accumulation in a mangrove forest and its relevance to local sea-level rise (Ilha Grande, Brazil). J. Coast Res. 242, 533–536. https://doi.org/10.2112/07-0872.1.

Sanders, C.J., Smoak, J.M., Naidu, A.S., Araripe, D.R., Sanders, L.M., Patchineelam, S.R., 2010a. Mangrove forest sedimentation and its reference to sea level rise, Cananeia, Brazil. Environ. Earth Sci. 60, 1291–1301. https://doi.org/10.1007/s12665-009-0269-0.

Sanders, C.J., Smoak, J.M., Naidu, A.S., Sanders, L.M., Patchineelam, S.R., 2010b. Organic carbon burial in a mangrove forest, margin and intertidal mud flat. Estuar. Coast Shelf Sci. 90, 168–172. https://doi.org/10.1016/j.ecss.2010.08.013.

Sanders, C.J., Smoak, J.M., Sanders, L.M., Sathy Naidu, A., Patchineelam, S.R., 2010c. Organic carbon accumulation in Brazilian mangal sediments. J. South Am. Earth Sci. 30, 189–192. https://doi.org/10.1016/j.jsames.2010.10.001.

Sasmito, S.D., Cameron, C., Taillardat, P., Clendenning, J.N., Friess, D.A., Murdiyarso, D., Hutley, L.B., 2019. Effect of Land-use and Land-cover Change on Mangrove Blue Carbon: a Systematic Review 1–12. https://doi.org/10.1111/gcb.14774.

Sasmito, S.D., Kuzyakov, Y., Lubis, A.A., Murdiyarso, D., Hutley, L.B., Bachri, S., Friess, D.A., Martius, C., Borchard, N., 2020a. Organic carbon burial and sources in soils of coastal mudflat and mangrove ecosystems. Catena 187. https://doi.org/10.1016/j.catena.2019.104414.

Sasmito, S.D., Sillanpää, M., Hayes, M.A., Bachri, S., Saragi-Sasmito, M.F., Sidik, F., Hanggara, B.B., Mofu, W.Y., Rumbiak, V.I., Hendri Taberima, S., Suhaemi Nugroho, J.D., Pattiasina, T.F., Widagti, N., Barakalla Rahajoe, J.S., Hartantri, H., Nikijuluw, V., Jowey, R.N., Heatubun, C.D., zu Ermgassen, P., Worthington, T.A., Howard, J., Lovelock, C.E., Friess, D.A., Hutley, L.B., Murdiyarso, D., 2020b. Mangrove blue carbon stocks and dynamics are controlled by hydrogeomorphic settings and land-use change. Global Change Biol. 26, 3028–3039. https://doi.org/10.1111/gcb.15056.

Semeniuk, V., 1994. Predicting the effect of sea-level rise on mangroves in Northwestern Australia. J. Coast Res. 10, 1050–1076. https://doi.org/10.2307/4298296.

Shannon, C., Weaver, W., 1963. The Mathematical Theory of Communication. University of Illinois Press. https://doi.org/10.1145/584091.584093.

Sharma, S., MacKenzie, R.A., Tieng, T., Soben, K., Tulyasuwan, N., Resanond, A., Blate, G., Litton, C.M., 2020. The impacts of degradation, deforestation and restoration on mangrove ecosystem carbon stocks across Cambodia. Sci. Total Environ. 706, 135416.

Sidik, F., Adame, M.F., Lovelock, C.E., 2019. Carbon sequestration and fluxes of restored mangroves in abandoned aquaculture ponds. J. Ind. Oce. Reg. 15, 177–192. https://doi.org/10.1080/19480881.2019.1605659.

Simpson, Edward H., 1949. Measurement of diversity. Nature 163, 688.

Smoak, J.M., Breithaupt, J.L., Smith, T.J., Sanders, C.J., 2012. Sediment accretion and organic carbon burial relative to sea-level rise and storm events in two mangrove forests in Everglades National Park. Catena 104, 58–66. https://doi.org/10.1016/j.catena.2012.10.009.

Soemodihardjo, S., Hardjowigeno, S., Naamin, N., Ongkosongo, O.S.R., Sudomo, 1991. Prosidings Seminar IV Ekosistem Mangrove, Bandar Lampung, 7-9 Agustus 1990. Panitia Nasional Program MAB Indonesia-LIPI, Jakarta.

Soper, F.M., MacKenzie, R.A., Sharma, S., Cole, T.G., Litton, C.M., Sparks, J.P., 2019. Non-native mangroves support carbon storage, sediment carbon burial, and accretion of coastal ecosystems. Global Change Biol. 25, 4315–4326. https://doi.org/10.1111/gcb.14813.

Sukardjo, S., Alongi, D.M., Ulumuddin, Y.I., 2014. Mangrove community structure and regeneration potential on a rapidly expanding river delta in Java. Trees. https://doi.org/10.1007/s00468-014-1021-2.

Tack, J.F., Polk, P., 1999. The influence of tropical catchments upon coastal zone: modeling the links between groundwater and mangrove losses in Kenya, India and Florida. In: Harper, D., Brown, T. (Eds.), Sustainable Management in Tropical Catchments. John Wiley and Sons Ltd., London, UK, pp. 359–372.

Tateda, Y., Nhan, D.D., Wattayakorn, G., Toriumi, H., 2005. Preliminary evaluation of organic carbon sedimentation rates in Asian mangrove coastal ecosystems estimated by ^{210}Pb chronology. Radioprotection 40, 527–532. https://doi.org/10.1051/radiopro.

Twilley, R.R., River-Monroy, V.H., Chen, R., Botero, L., 1998. Adapting and ecological mangrove model to stimulate trajectories in restoration ecology. Mar. Pollut. Bull. 37, 404–419.

Villa, J.A., Mitsch, W.J., 2014. Carbon sequestration in different wetland plant communities in the Big Cypress Swamp region of southwest Florida. Int. J. Biodivers. Sci. , Ecosyst. Serv. Manag. 37–41. https://doi.org/10.1080/21513732.2014.973909.

Watanabe, K., Seike, K., Kajihara, R., Montani, S., Kuwae, T., 2019. Relative sea-level change regulates organic carbon accumulation in coastal habitats. Global Change Biol. 1–15. https://doi.org/10.1111/gcb.14558.

White, P.S., Jentsch, A., 2001. The Search for Generality in Studies of Disturbance and Ecosystem Dynamics, vol. 62, pp. 399–450. https://doi.org/10.1007/978-3-642-56849-7_17.

Woodroffe, C.D., Mulrennan, M.E., Chappell, J., 1993. Estuarine infill and coastal progradation, southern van Diemen Gulf, northern Australia. Sediment. Geol. 82, 257–275.

Woodroffe, C.D., Rogers, K., McKee, K.L., Lovelock, C.E., Mendelssohn, I.A., Saintilan, N., 2016. Mangrove sedimentation and response to relative sea-level rise. Ann. Rev. Mar. Sci. 8, 243–266. https://doi.org/10.1146/annurev-marine-122414-034025.