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Soil and above-ground carbon stocks in a planted tropical mangrove forest (Can Gio, Vietnam)

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Abstract

Can Gio mangrove is the largest in Vietnam, developing on approximately 35000 hectares. This forest was partially destroyed during the Vietnamese war. A restoration program was developed between the late 70s and the early 90s, using *Rhizophora apiculata* Blume propagules. Currently, the Can Gio mangrove forest regenerates naturally and presents a specific species zonation along the intertidal elevation gradient. *Rhizophora* dominates the inner forest at the highest elevation, while at an intermediate location, *Rhizophora* and *Avicennia* cohabit with other scattered species. The lowest position is colonized by *Avicennia*. Within this context, the main objectives of this study were to determine the soil physicochemical characteristics, as well as the quality (C/N ratios and $\delta^{13}C$) and the quantity (carbon content and stocks) of the organic matter stored beneath each mangrove stand. In addition, we were interested in determining the above-ground biomass and the total carbon stocks of the ecosystem (without considering the below-ground biomass). Carbon stocks of the Can Gio mangrove forest ranged from 150 to 479 Mg C ha^{-1}, with up to 86 % of the C stored in the upper meter of the soil. The inner forest has the highest stock, followed by the transitional forest, and the fringe forest. The depth extension of the root system of the current forest was estimated, and its contribution to the soil carbon stock was calculated, using the adjacent mudflat as a proxy for the antecedent stocks. Our results show that, for the last 40 years, the current mature planted *Rhizophora* forest stored 25.26 Mg C ha^{-1}. Consequently, mangrove plantation and restoration after the war was a success in terms of carbon storing. We suggest that the destruction of the Can Gio mangrove forests for urban development would induce the loss of an efficient CO_2 sink.

1. Introduction

Mangroves forests cover around 137760 km² between 30° N and 40° S, with the largest percentage of their surface observed between 20° N and 20° S (Giri et al., 2011). Mangroves provide significant ecosystem services such as habitat for various terrestrial and marine animals that are critical for the coastal biodiversity in the tropics (Alongi, 2008; Mumby et al., 2004; Nagelkerken et al., 2008; Saenger, 2002), reef against erosion and natural disasters (Alongi, 2008; Barbier, 2006), or even as a trap for suspended and contaminant materials (Kathiresan, 2004; Rivera-Monroy et al., 1999). In addition, mangroves strongly contribute to the economic growth of emerging countries (Mukherjee et al., 2014), providing charcoal, firewood, and construction materials for local communities.

Due to their high productivity (average of 218 ± 72 Tg C yr^{-1}) (Bouillon et al., 2008), the anoxic character of their soils that limits organic matter degradation (Kristensen et al., 2008), their high storage capacity (Breithaupt et al., 2012) and their global distribution, mangroves play a key role in carbon cycling in the coastal ocean (Kauffman et al., 2011). Mangroves can store carbon both in their biomass and soils; however, this ability depends on many parameters and may be highly variable. Their biomass varies notably according to latitude with higher biomass under the tropics (Saenger and Snedaker, 1993), climate with higher biomass under the wet than under the dry tropics (Adame et al., 2013), and even the age of the forest (Fromard et al., 1998). In addition, a large part of the carbon content may be stored in their soil, with up to 98 % and up to 90 % for estuarine and coastal mangroves, respectively (Donato et al., 2011). Their potential soil carbon storage was recently estimated at 1023 Mg C ha^{-1} (Donato et al., 2011), much higher than other highly productive ecosystems, such as rain forests (218.6 Mg C ha^{-1}), peat swamps (370.1 Mg C ha^{-1}), or salt marshes (537.8 Mg C ha^{-1}) (Alongi, 2014). However, mangrove soil carbon stocks depend also on various parameters, such as the latitudinal position of the mangrove, the tree species developing at the surface, soil salinity, and even nutrient availability (Adame et al., 2013; Alongi, 2002; Jacotot et al. 2018; Kauffman et al., 2011; Rahman et al., 2015; Sanders et al., 2010; Wang et al., 2013). Particularly, their position within the tidal zone or along an estuary appears to be critical to their potential soil carbon storage capacities. For example, in an estuarine mangrove forest developing in Mexico, Adame et al. (2015) reported soil carbon stocks ranging from 744 to 912 Mg C ha^{-1} for the upper estuary and from 537 to 1115 Mg C ha^{-1} for the lower estuary.

Recently, Vietnam became the first Asian country to implement a national program of payment for forest environmental services (PFES). In Vietnam, mangroves cover around 270000 ha (FAO, 2015) with more than 80 % distributed in the southern part of the country, below 10° N latitude (Hawkins et al., 2010). The Can Gio mangrove, located between Ho Chi Minh City (HCMC) and the South China Sea (Bien Dong in Vietnamese), is actually the largest contiguous mangrove in Vietnam with a surface of around 35000 ha. During the Vietnamese war (1964-1971), approximately 57% of this mangrove was destroyed by the spraying of a mixture of herbicides and defoliants (Ross, 1975). After the war, in the late 70s and early 90s, a vast reforestation program was implemented by the HCMC Forest Department, and most of the Can Gio area was replanted using *Rhizophora apiculata* Blume propagules collected in the Mekong Delta. In January 2000, the mangrove forest of Can Gio was registered in the UNESCO World Network of Biosphere Reserves list. In addition, the environmental value of the Can Gio mangrove forest was taken into account in the National Strategy on Climate Change and Sea Level Rise by the Vietnamese government. Currently, the Can Gio mangrove forest is highly diversified, with a total of 77 mangrove species (35 true mangroves and 42 associates) (Tri et al., 2000), whose main species present a specific zonation along the intertidal elevation gradient. This zonation includes, from the lowest to the highest position: i) a fringe forest that is dominated by *Avicennia alba*; ii) a transitional forest, composed of a mixture of *R. apiculata A. alba*, *A. officinalis* and sparse *Excoecaria agallocha* and *Sonneratia alba*; and iii) an inner forest, which is mainly composed of mature *R. apiculata.* It was recently demonstrated that the mineralization of mangrove-derived organic matter has a key role in sustaining coastal food webs in the Can Gio estuary (David et al. 2018a, David et al. 2019). Recent studies also showed that the total carbon stocks in the Can Gio mangrove forest could reach 1000 Mg C ha^{-1} (Dung et al., 2016; Nam et al., 2016) and that the below-ground carbon accumulation rates may reach 3.24 Mg C ha^{-1} yr^{-1} (MacKenzie et al., 2016). However, these studies were interested neither in soil organic carbon (SOC) quality nor in the influence of soil elevation on carbon stocks of the different stands.

Therefore, the main objectives of this study were: i) to evaluate the soil physicochemical parameters, the organic matter quality, and the vegetation characteristics of three different mangrove stands and of the adjacent mudflat developing along the elevation gradient, ii) to determine the carbon stocks in the biomass and in the first meter of soil, and iii) to calculate the soil carbon stocks related to forest plantation. We hypothesized that carbon stocks in the soil and in the biomass depend on the mangrove species and, therefore, to the position of the stand along the intertidal elevation gradient. To reach our goals, core samples were taken within each of the four stands. Supplementary cores were also collected in the adjacent mudflat to serve as a soil reference (i.e., before the colonization of the soil by the mangrove). Then, physicochemical parameters (pH, redox, and salinity) and the distribution of C/N ratios and δ^{13}C stable isotopes with depth in each stand were measured. Carbon stocks were determined by combining bulk density and total organic carbon content for the soil and by using biomass measurements with specific (Vinh et al., 2019) and generic allometric equations for the above-ground stocks.

2. Methods

2.1. Study site

The present study was conducted in the Can Gio mangrove forest in Southern Vietnam (Fig. 1a). This mangrove forest of around 35000 ha is the largest contiguous mangrove in Vietnam. Located in the district of Can Gio, one of the 24 districts of Ho Chi Minh City, the Can Gio mangrove forest is at the deltaic confluence of the rivers Sai Gon, Dong Nai, and Vam Co, flowing into the East Sea (Bien Dong in Vietnamese). The Can Gio mangrove forest is composed of several species that follow a specific zonation determined by soil elevation. From the lowest to the highest intertidal zone, the forest is composed of a mudflat that is totally denuded of vegetation; a fringe forest, dominated by *Avicennia* spp. (mainly *A. alba*); a transitional forest, mainly composed of *R. apiculata*, *A. alba*, *A. officinalis*, as well as sparse *Excoecaria agallocha* and *Sonneratia alba*; and an inner forest dominated by mature *R. apiculata*.

The climate in this region is typically monsoonal (type Am in Köppen-Geiger classification) with two main seasons: a wet season from May to November and a dry season from December to April. The annual average rainfall is 1816 mm, spread over 154 rainy days, with approximately 80 % of the rainfall occurring during the wet season. As a result, the flow of sediments transported by the river crossing the mangrove forest during the wet season is elevated (~160 million tons (Milliman and Meade, 1983)), leading to the accretion of the deltaic plain downstream. Conversely, during the dry season, the fresh water flow is strongly reduced, and the mangrove forest is severely affected by saline intrusions. The Can Gio estuary is characterized by irregular semi-diurnal tides with a tidal range of 2 to 4 m.

2.2. Field measurements and carbon stocks

2.2.1 Elevation

The elevation of each stand relative to mean sea level (MSL) was measured with a dye-type tide gauge (Clough, 2014; English et al., 1997; Schmitt and Duke, 2014). First, the cotton tape was soaked in a water soluble food dye and attached to a wooden stake of about 2.5 m in length. At low tide, the wooden stakes were inserted deeply into the soil starting from the river edge. The distance between the two columns was 2 m in the mudflat, 5 m in the fringe forest, and then every 10 m until the inner forest. After high tide, we measured the height of the washout line above the ground (Albers and Schmitt, 2015; Clough, 2014). In this study, data from the Vung Tau tide station, Viet Nam (10.3333° N, 107.0667° E) were used to report our data relative to MSL in this zone. The MSL at Vung Tau tide station in 2016 was 2.16 m. To determine MSL at the study site, the following equation was used: MSL = highest tide value – MSL at Vung Tau. Monthly and annual mean sea levels and the daily tide series are collected and published by the Permanent Service for Mean Sea Level (Pissierssens, 2002).

2.2.2. Cores collection and soil physicochemical parameters

Soil cores were collected in triplicate with a gouge auger (1 m long, 3 cm wide) attached to a cross handle. Cores were collected during the dry season of 2016 and the wet season of 2017 from the surface to a depth of 100 cm in each mangrove stand as S1 in the mudflat, S2 in the fringe forest, S3 in the transitional forest, and S4 in the inner forest (Fig. 1c). The auger was carefully inserted into the soil to minimize disturbance of the core surface, twisted, and finally removed, following the method used by MacKenzie et al. (2016). Each core was then separated into different depth intervals: 0 – 2.5 cm, 2.5 – 5 cm, 5 – 7.5 cm, 7.5 – 10 cm, 10 – 15 cm, 15 – 20 cm, 20 – 25 cm, 25 – 30 cm, and then every 10 cm from 30 to 100 cm. For each interval, one subsample of a known volume was collected and placed in a zippered bag that was immediately sealed in aluminum foil to minimize gas exchange and stored in a frozen box until it reached the laboratory. All samples were, then, dried by freeze-drying at -52 °C until a constant weight was achieved. Finally, the dry bulk density (DBD) of each sample was determined by dividing its dry mass by its fresh volume.

Additional cores were collected to measure pore-water salinity, redox potential (Eh), and pH. Pore-water salinity was measured with a hand-held refractometer. Redox potential and pH were measured using, respectively, a combined Pt-Ag/Ag-Cl electrode and a glass electrode, both connected to a pH/mV/T meter (Marchand et al., 2004). The pH electrode was calibrated prior to sampling using three standard solutions of pH 4, 7, and 10 at 25 °C (National Institute of Standards and Technology, USA), and the redox electrode was checked prior to utilization with a 0.43 V standard solution and demineralized water (Marchand et al., 2004; Thanh-Nho et al., 2017).

2.2.3. Soil TOC, TN, δ^{13}C, and soil carbon stocks

Soil total organic carbon (TOC), total nitrogen (TN), and δ^{13}C values were determined using an elemental analyzer coupled to an isotope ratio mass spectrometer (Integra2, Sercon, UK). The analytical precisions of the analyzer were checked using the IAEA-600 caffeine standard (IAEA Nucleus) and were less than 1 % for TOC, 0.15 % for N, and 0.3 % for δ^{13}C. All the analyses were performed at the French Institute for the Sustainable Development (IRD) of Noumea, New Caledonia, France. The δ^{13} C values are reported permil (‰) deviations from a Pee-Dee Belemnite (PDB) limestone carbonate as the standard using the following equation:

$$\delta^{13}\text{C (‰)} = \left(\frac{\frac{^{13}C}{^{12}C}_{sample}}{\frac{^{13}C}{^{12}C}_{standard}} - 1 \right) \times 1000 \qquad \text{(Eq. 1)}$$

The soil carbon stocks of each mangrove stand (Mg C ha^{-1}) were determined using the following equation:

$$\text{Soil carbon stock (Mg C ha}^{-1}\text{)} = \text{TOC (\%)} * \text{DBD (g cm}^{-3}\text{)} * \text{depth interval (cm)} \qquad \text{(Eq. 2)}$$

2.2.4. Above-ground carbon stocks

In November 2016, three transects (150 m long and 20 m wide) were established following an elevation gradient in the mangrove forest (Fig. 1c). All transects encompassed the four different stands of the mangrove (i.e., the mudflat, the fringe, the transitional, and the inner forests). All trees within the transects were counted and their DBH at 1.3 m above the soil were measured, except for *R. apiculata*, whose diameters were measured just above the highest prop root.

For *R. apiculata,* the above-ground biomass was calculated using the allometric equation specifically developed for this species in Southern Vietnam (Vinh et al., 2019):

$$WR_{Total} = 0.38363 * DBH^{2.2348} \qquad \text{(Eq. 3)}$$

For the other species, the above-ground biomass was determined using the common allometric equations developed by Komiyama et al. (2005).

$$W_{Total} = 0.251*DBH^{2.46} \qquad \text{(Eq. 4)}$$

where, in Eq. 3, WR_{Total} is the total above-ground biomass of *R. apiculata* (kg), and D is the diameter above the highest prop root of *R. apiculata* (cm); and in Eq. 4. W_{Total} is the total above-ground biomass biomass of the other species (kg), and DBH is the diameter at breast height of the other species (cm).

Above-ground carbon stocks were then estimated by multiplying the above-ground biomass by a carbon conversion factor. Two conversion factors were used in this study: 0.4409 for *R.apiculata*, which is the specific carbon conversion factor for this species (Vinh et al., 2019), and 0.451 for the other species (Hiraishi et al., 2014). Eventually, the total above-ground carbon stocks per area (Mg C ha^{-1}) were calculated by summing the above-ground carbon stock of *R. apiculata* and the one for the other trees using the tree density per hectare.

2.3. Statistical analyses

A parametric two-way analysis of variance (ANOVA) was used to test the significant effects ($p < 0.05$) of seasons and sites on Eh, pH, and pore-water salinity. A two-way ANOVA was also applied to assess the significant differences ($p < 0.05$). A Student’s t-test was used to test significant differences ($p < 0.05$) between $\delta^{13}C$ values, C/N ratios, and C concentrations of an upper layer from 0 to 40 cm and a lower layer from 40 to 100 cm, and for Eh and pH for the transitional forest and inner forest. All statistical analyses were performed using XLSTAT software version 2017.4 for Mac OS10.13.4.

3. Results

3.1. Can Gio mangrove distribution

The relative elevations of the soil surface at the four sampling stations increased from the river toward the inner forest. The *A. alba* forest developed between +20 and +30 cm above MSL, the transitional forest between +30 and + 40 cm, and the *Rhizophora* stand between +40 and + 75 cm above MSWL (Fig. 2).

The fringe forest had a mean DBH value of 10.7 cm, a tree density of 1327 ind ha^{-1}, and a basal area of 11.9 m^2 ha^{-1}. The latter parameter increased to the inner forest with a value reaching 28.6 cm; the inner forest was also characterized by the highest mean DBH, 18.6 cm. The transitional forest was characterized by the highest tree density, 3727 individuals per hectare (ind ha^{-1}), and the lowest DBH, 8.5 cm (Table 1).

3.2. Soil physicochemical parameters (pore-water salinity, Eh, and pH)

Pore-water salinity remained relatively stable along the core profiles during both seasons in the mudflat as well as in the fringe and the transitional forests, with values ranging from 10.0 to 23.3, 18.0 to 28.3, and 12.0 to 30.0, respectively (Fig. 3 a, d, g). However, in the inner forest, pore-water salinity increased from 10.0 and 27.3 at the soil surface to 18.0 and 31.7 at 20 cm depth, during the dry and the wet season, respectively. Below 20 cm, pore-water salinity remained stable until 100 cm of depth (Fig. 3 l). When integrating the entire sampled profile, the season had a significant effect ($p < 0.001$) on pore-water salinity, with higher values during the dry season in all stands, with the exception of the transitional forest where no significant differences ($p > 0.05$) in mean pore-water salinity were observed. In addition, pore-water salinity was significantly different between the four zones in both seasons ($p < 0.001$). Mean values were 20.07 ± 1.78 and 12.47 ± 3.48 in the mudflat, 24.8 ± 1.8 and 19.8 ± 1.3 in the fringe forest, 19.0 ± 1.8 and 19.5 ± 3.5 in the transitional forest, and 29.5 ± 2.8 and 16.3 ± 4.5 in the inner forest, for the dry and the wet season, respectively. Consequently, pore-water salinity increased along the intertidal elevation gradient with lower values at the lowest position and the higher values in the higher position.

In the four zones, Eh values decreased with depth, from 150 to -50 mV in the mudflat, from -50 to -250 mV in the fringe and the transitional forests and from -50 to -400 mV in the inner forest (Fig. 3 b, e, h, m). When integrating the whole cores, from the top to the bottom, at 100 cm depth, Eh significantly ($p < 0.001$) decreased from the mudflat to the inner forest. Mean values were -10.7 ± 104.9 and 46.6 ± 90.9 mV in the mudflat, -52.2 ±70.0 and -107.9 ± 84.1 mV in the fringe forest, 44.7 ± 104.5 and -118.9 ± 93.6 mV in the transitional forest, and -268.9 ± 185.9 and -117.4 ±57.2 mV in the inner forest, for the dry and the wet season, respectively. However, although Eh was higher during the dry season than during the wet season, these differences were not significant in any of the stands ($p > 0.05$, Table 2). Finally, Eh values in the upper layer (0 – 40 cm) were significantly different ($p < 0.05$) from the lower layer (40 – 100 cm) for both the transitional forest and inner forests.

Concerning pH, values did not vary significantly during both seasons in the mudflat, with values ranging from 6.9 to 7.5 (Fig. 3c, f, k and n). For the other stands, pH values varied with depth and with season, from 6.2 to 7.6. When integrating the whole core (i.e., from the soil surface to 100 cm depth), pH values were significantly different between stands ($p < 0.001$), with higher mean values in the mudflat, followed by the fringe and the transitional forests, and with the lowest mean value in the inner forest. In addition, there were significant differences ($p < 0.05$) between the upper layer (0 – 40 cm) and the lower layer (40 – 100 cm) for both the transitional and inner forests.

3.3. Soil organic matter characteristics (DBD, TOC, TN, $\delta^{13}C$, and C/N ratios)

For all stands, DBD values remained stable along the soil profile, from the surface to 100 cm depth. However, mean DBD values for the first meter of soil were statistically different ($p < 0.001$) between each stand. The mean DBD values were 0.52 ± 0.06, 0.62 ± 0.02, 0.63 ±

0.03, and 0.64 ± 0.08 g cm^{-3} in the mudflat, the fringe, the transitional, and the inner forests, respectively.

In the mudflat and in the fringe forest, TOC values were stable from the soil surface to the bottom of the core, with mean values of 2.42 ± 0.34 and 3.2 ± 0.27 %, respectively (Fig. 4 a). In the transitional and the inner forests, TOC values between the upper (0 – 40 cm) and the deep layers (40 – 100 cm) were significantly different ($p < 0.05$). Mean TOC values were 4.14 ± 1.05 and 7.03 ± 3.10 % for the transitional forest and 4.36 ± 1.31 and 6.75 ± 2.80 % for the inner forest for the upper and the deep layers, respectively (Fig. 4 a). In addition, the average TOC values for the first meter of soil were significantly different ($p < 0.001$) between the four zones.

C/N ratios were relatively stable throughout the entire sampled soil profile in the mudflat and in the fringe forest with values ranging from 11.2 to 12.1 and from 11.2 to 15.7, respectively (Fig. 4 b). However, similar to TOC, C/N ratios between the upper 40 cm of soil in the transitional and inner forests were statistically different ($p < 0.05$) when compared to the lower 60 cm. The mean values of C/N ratios were 12.2 and 18.2 for the transitional forest and 17.5 and 25.7 for the inner forest for the upper and the lower layers, respectively (Fig. 4 b). In addition, when considering the complete sampled profile, C/N ratios were significantly different ($p < 0.001$) between the four stands and increased landward. The mean values were 11.61 ± 0.22, 12.08 ± 1.09, 14.44 ± 4.53, and 22.78 ± 5.49 for the mudflat, the fringe forest, the transitional forest, and the inner forest, respectively.

Concerning $\delta^{13}C$, in the mudflat and in the inner forest, the values slightly increased from -30 ‰ and -32 ‰ at the soil surface to -27 ‰ and -31 ‰ at 20 cm depth and then remained relatively stable until the bottom of the cores (Fig. 4 c). In the fringe forest, $\delta^{13}C$ values rapidly dropped from -27 ‰ at the soil surface to -32 ‰ at 10 cm depth, and then gradually increased to a mean value of -29‰ from 20 to 80 cm depth. However, a second rapid drop to -34 ‰ was observed in the profile at 90 cm depth (Fig. 4 c). Finally, in the transitional forest, $\delta^{13}C$ values decreased from -29 ‰ at the soil surface to -30 ‰ at 25 cm depth and then remained stable until 100 cm depth (Fig. 4 c). When integrating the entire core, the mean $\delta^{13}C$ were significantly different between the four zones ($p < 0.001$), with mean values of -28 ± 0.8 ‰, -30 ± 2.3 ‰, -29 ± 1.4 ‰, and -31 ± 0.3 ‰ in the mudflat, the fringe, the transitional forest, and the inner forests, respectively.

3.4. Carbon stocks in the above-ground biomass and in the soils of the different stands

Above-ground carbon stocks were estimated at 24.3 ± 5.1 Mg C ha^{-1} for the fringe forest, 91.7 ± 29.4 Mg C ha^{-1} for the transitional forest, and 118.8 ± 9.5 Mg C ha^{-1} for the inner forest. In addition, above-ground carbon stocks were significantly different between the different stands ($p < 0.001$). No above-ground biomass was measured in the mudflat due to the absence of vegetation in this zone. Soil carbon stocks increased with the elevation gradient, with higher values in the inner forest (360.4 Mg C ha^{-1}), followed by the transitional forest (219.8 Mg C ha^{-1}), the fringe forest (157.6 Mg C ha^{-1}), and the mudflat (150.2 Mg C ha^{-1}) (Fig. 5). In addition, soil carbon stocks were significantly different between the different stands ($p < 0.001$). Eventually, the total carbon stocks (without the below-ground biomass) were 150.2 ± 19.6, 181.9 ± 24.9, 311.5 ± 28.1, and 479.2 ± 32.6 Mg C ha^{-1}, for the mudflat, the fringe, the transitional forest, and the inner forest, respectively (Fig. 5).

4. Discussion

4.1. Mangrove zonation in Can Gio

Between 1978 and 1994, a vast mangrove reforestation program was undertaken with *R. apiculata* as a primary species (Hong and San, 1993). When *Rhizophora* stands were established, the mangrove area extended through natural regeneration and rapid colonization of mudflats along riverbanks notably by *A. alba*, which can be considered as a pioneer species (see for example Balke et al. (2011); Brunt and Davies (2012); Naidoo and Naidoo (2017); Proisy et al. (2009)) (Fig. 2 a, b). This fringe forest developed at lower elevations as compared to the *R. apiculata* forest and was separated from the latter by a transitional forest composed of several species. Mangrove zonation often manifests itself as a mosaic that varies according to physical, biological, and chemical interactions established between plant and substrate in a given area. Pore-water salinity was often considered as the main driver of the zonation (Banerjee et al., 2013; Marchand et al., 2012; 2011), because mangrove plants have different abilities to cope with this factor (Ellison, 1998; Mckee, 1993; Walsh, 1974). For instance, under semi-arid climate, pore-water salinity increased landward, where evaporation processes were intense due to the rare periods of immersions and low precipitation and could reach to a value greater than 50. As a consequence, *Avicennia* trees, which can cope with high salinity (Kendall and Skipwith, 1969; Khan and Aziz, 2001; Marchand et al., 2004; Ukpong, 1997), developed at higher tidal position than the *Rhizophora* trees, which colonized the seaward zone of the mangrove forest. In Can Gio, the zonation was the opposite because pore-water salinity never reached such high values. In fact, pore-water salinity increased from the fringe forest to the interior forest due to its higher elevation that induced more evaporation, but it was never higher than 30. The Can Gio mangrove forest is an estuarine mangrove with freshwater inputs from the Sai Gon and the Dong Nai Rivers. Along the estuary, salinity values ranged from 2 to 26 during the year; and even at the mouth of the estuary, salinity never reached the value of seawater, possibly due to the high freshwater inputs from these two rivers and also from the Mekong delta, which is further south of the study site (David et al., 2018b; Thanh-Nho et al., 2018). Furthermore, during the rainy season, the intense rainfall brought additional freshwaters that induce a dilution of pore-water salinity, with mean values for the mature *Rhizophora* forest decreasing from 29 to 16. In the latter stand, salinity also increased with depth, notably because dilution with rainwater occurred in the upper sediment during the rainy season, but also possibly because dissolved salts could migrate at depth through convection processes and accumulate there (Marchand et al., 2004).

4.2. Influence of mangrove development on soil properties

Organic carbon (OC) content in mangrove sediments usually ranges from 0.5 % to 15 %, with a median value of 2.2 % (Kristensen et al., 2008). In the Can Gio forest, soil OC increased with increasing elevation landward, ranging from 2.42 % in the low intertidal zone to 5.82 % in the high intertital zone, indicating that the interior forest had accumulated a larger organic carbon stock. The interior forest studied here is composed of mature *Rhizophora* trees, which were planted in 1978. Therefore, mangrove-derived organic matter had accumulated in its soil for almost 40 years at the date of the coring. *Rhizophora* trees under tropical climates as in southern Vietnam are highly productive (Alongi, 2014), which can positively increase soil carbon stocks. In addition, this stand was the furthest from the tidal creek; and as a consequence, tidal flushing of leaf litter was reduced. Leaf litter can accumulate and increase the soil carbon stocks. Conversely, due to their more recent development, lower productivity, and proximity to tidal creeks, the soils of the fringe stand composed of *A. alba* contained less organic carbon than the mature inland *Rhizophora* forest.

These gradients of elevation, length of tidal immersion, and carbon content strongly influence redox conditions and pH of the soil. Redox potential (Eh) decreased with increasing elevation from the mudflat to the mature *Rhizophora* stand. Additionally, within the *Rhizophora* stand, the redox condition became rapidly anoxic with depth (Fig. 3). We suggest that the higher organic content beneath this stand induced a higher electron acceptor demand, which were less renewed by the tides due to its high position in the tidal zone. The higher redox values measured beneath the *Avicennia* stands may also be related to the ability of this mangrove species to aerate the sediment through its root system as described by Scholander et al. (1955). Hesse (1961) also observed that *Rhizophora* soils were anoxic most of the time and sulfidic. This difference between the two species was later confirmed in different countries and was suggested to be related to different organic enrichment of the soil, specific abilities of the root system, and different positions in the tidal zone (Marchand et al., 2004; Marchand et al., 2011; Mckee, 1993). pH values also varied along the elevation gradient of the studied tidal zone, decreasing from 6.8 in the mudflat to 6.2 in the mature *Rhizophora* stand. We suggest that the increased organic content and its decay processes were responsible for this soil acidification. In addition, in anoxic mangrove soils, sulfides minerals could precipitate (Balk et al., 2016), and slight modifications of the redox conditions could induce their oxidation, which could result in soil acidification (Noel et al., 2017; 2014).

4.3. Characterization of soil organic matter with depth and along the intertidal elevation gradient

In mangrove soils, organic matter is usually a mixture between autochthonous organic matter (leaf litter, roots debris, and microphytobenthos) and allochthonous organic matter derived from marine and/or terrestrial origins (Kristensen et al., 2008). However, in a distinct area, the respective contribution of each source in the carbon pool depends on several parameters, including trees productivity, mangrove position, tidal range, freshwater inputs, position of in the intertidal zone, etc. In the Can Gio mangrove soils, $\delta^{13}C$ and C/N ratios ranged between -28 ‰ to -31 ‰ and between 12 to 22, respectively, and were consistent with those previously reported (Bouillon et al., 2003; Prasad et al., 2017, Jacotot et al., 2018). However, OM quality differed along the intertidal elevation gradient, with higher C/N ratios and depleted $\delta^{13}C$ values as the elevation increased. These gradients suggested an increased contribution in the upper intertidal zone of mangrove-derived organic matter, characterized by elevated C/N ratios and depleted $\delta^{13}C$ values (Bosire et al., 2005; Jennerjahn and Ittekkot, 1997; Kristensen et al., 2008; Marchand et al., 2005). These results were consistent with our previous hypothesis of a higher enrichment of the soil resulting from mangrove development and a greater contribution of *Rhizophora* leaf litter in the inner forest due to the higher productivity of the stand, the limited tidal export, and the anoxic character of the soils that limits OM decay process and favors its accumulation. Conversely, towards the tidal creek side of the mangrove, the enriched $\delta^{13}C$ and lower C/N ratios, close to 12, suggested a greater contribution of phytoplankton or phytobenthos and/or more degraded higher plant debris. In a recent study (Vinh et al., 2020), we showed that leaf litter in the *Avicennia* stand was more rapidly decomposed, and that decay rates were even enhanced during the monsoon. Consequently, we suggest that the position of the stand along the intertidal elevation gradient and its species composition influenced the organic matter characteristics of the soil.

Surprisingly, an organic-rich layer was observed at depth beneath the transitional and the inner forests from 40 to 100 cm depth. In the mangroves of southern Vietnam, MacKenzie et al. (2016) determined an average vertical accretion rate of around 0.99 ± 0.09 cm yr^{-1}. This vertical accretion rate was relatively elevated and reflected notably the high sedimentation rate characterizing most Asian estuaries. During the wet season, strong rainfalls induce high erosion rates in the upper watersheds. As a result, high quantity of sediments are transported by the rivers and deposited along the coastlines, notably in mangrove forests. Following the accretion rate determined by MacKenzie et al. (2016), the upper layer observed in this study (from 0 to 40 cm depth) would have started to be deposited 40 years ago, which almost corresponds to the beginning of the reforestation program started in 1978 (Hong and San, 1993). Consequently, we suggest that the upper layer corresponds to the development of the current forest (0 – 40 cm), and the lower layer (40 – 100 cm) accumulated before mangrove destruction during the Vietnamese war. Interestingly, this buried layer was at least 50 % enriched in carbon compared to the upper layer for the same sediment thickness and was also characterized by higher C/N ratios (9.7 vs. 22.3 and 14.0 vs. 37.1, for the upper and the lower layers, respectively) and depleted δ^{13}C values (-30.7‰), which suggested that the former forest was more productive and/or accumulated organic carbon during a long period.

4.4. Influence of mangrove development on carbon stocks

Carbon stocks in the above-ground biomass were 118.8 ± 9.5 Mg C ha^{-1} for the inner forest, 91.7 ± 29.4 Mg C ha^{-1} for the transitional forest, and 23.3 ± 5.1 Mg C ha^{-1} in the fringe forest. Differences between stands may be related to the age of the forests, as the inner forest was planted 40 years ago, while the other stands naturally regenerated recently. In addition, *Rhizophora* trees that dominates the inner forest and extends into the transitional forest are generally more productive than *Avicennia* ones (Komiyama et al., 2008) that colonize the fringe forest. Nevertheless, these results were in the range of those previously observed from other studies in Southern Vietnam, with values ranging from 13.4 to 210.7 Mg C ha^{-1} (Dung et al., 2016; Tue et al., 2014). However, the above-ground carbon stocks in Can Gio were much lower than those in other mangrove forests. For example, in Malaysia, the above-ground carbon stocks reached 202.9 Mg C ha^{-1} (Putz and Chan, 1986). This difference may be explained by the age of the forest, as the forest in Malaysia has developed for 80 years—twice the age of the one in Can Gio, and by silvicultural activities (Vinh et al., 2019).

In Can Gio, soil carbon stocks ranged from 150 to 360 Mg C ha^{-1}, which was consistent with and even higher than the reported values for other mangrove forests in Vietnam, ranging from 144 to 233 Mg C ha^{-1} (Dung et al., 2016; Tue et al., 2014). However, these values were lower than the ones of tropical mangroves that ranged from 337 to 640 Mg C ha^{-1} (Adame et al., 2013; Castillo et al., 2017; DelVecchia et al., 2014; Hossain, 2014; Kauffman et al., 2011). We suggest that partial destruction of the mangrove during the Vietnamese War may have prevented organic matter to accumulate for a while and probably allowed the existing material to be eroded by tides and freshwater circulation, explaining these low values for soil carbon stocks. Soil carbon stocks in the mangrove of Can Gio represented between 70 to 86 % of its total carbon stocks (without the below-ground biomass) and was consistent with other Indo-Pacific mangrove forests (Donato et al., 2011; Kauffman et al., 2011; Liu et al., 2014).

Recently, some authors suggested that integration depth is of major concern when determining carbon stocks in mangrove forests (Lunstrum and Chen, 2014; Marchand, 2017; Jacotot et al., 2018). In Can Gio, integrating the soil carbon stocks down to one meter takes into account the stocks linked to the development of the current forest, replanted after the Vietnamese war as discussed above, and a part of the stocks that accumulated before the destruction of the forest. When considering only the development of the current forest (i.e., 0 – 40 cm, assuming a sedimentation rate of 0.99 ± 0.09 cm yr^{-1} as discussed above), soil carbon stocks were much at 85.7 ± 33.92 Mg C ha^{-1} in the inner forest. We did not calculate this stock for the other stands considering that it was a natural regeneration that occurred later than the planting and had an OM quality depth profile different from the mature *Rhizophor*a forest. In addition, knowing the age of the forest and the amount of carbon stored in its soil since its development allowed the precise contribution of this forest to the enrichment in soil organic matter to be determined. Doing this, the antecedent carbon stocks (i.e., the stocks that were present before the apparition of the forest) must be determined (Lal, 2005). In our study, the stocks in the mudflat were chosen as a proxy for the antecedent carbon stocks. Therefore, for the last 40 years, the actual forest contributed to an enrichment of 25.26 Mg C ha^{-1} in the inner forest. As a result, the carbon burial rate in the mature *Rhizophora* was ~0.6 Mg C ha^{-1} yr^{-1}, which is lower than the global value of 1.35 Mg C ha^{-1} yr^{-1} reported by Bouillon et al. (2008). However, our result may be underestimated, considering that the mudflat was probably enriched by mangrove-derived organic matter, as suggested by the depleted $\delta^{13}C$ values and the high TOC content. Nevertheless, this study demonstrated that increasing mangrove areas by either restoration or expansion is an important way of increasing carbon storage in the coastal ocean, and mangroves must be considered in future climate change mitigation programs.

5. Conclusions

Degraded by the spraying of defoliants during the Vietnam War, mangrove forests in Can Gio Estuary successfully recovered through replantation and natural regeneration. They now store a high amount of carbon both in their biomass and in their soils. Their destruction for infrastructure development along the coastline would result in the loss of an efficient CO_2 sink.

The main conclusions of this study can be summarized as follow:

1. C stocks in the above-ground biomass were significantly different between stands, increasing landward, reaching up to 118.8 ± 9.5 Mg C ha^{-1} for the mature *Rhizophora* stand. Differences in carbon stocks in the above-ground biomass between stands were related to different forest ages, mangrove species, and tree densities, the latter being managed by thinning.

2. The specific zonation, with planted *Rhizophora* trees at the highest elevation in the tidal zone (with a limited pore-water salinity value due to a monsoon-dominated climate), and natural colonization of the river banks by *Avicennia* trees resulted in gradients in the soil physicochemical properties from the mudflat to the inner forest. Due to their more recent development, lower productivity, and proximity to tidal creeks, the soils of the fringe stand composed of *A. alba* contained less organic carbon than the mature inland *Rhizophora* forest. These gradients of elevation and of carbon content strongly influenced the redox conditions and pH in the soil; with both decreasing from the mudflat to the mature *Rhizophora* stand.

3. Regarding soil organic matter quality, the $\delta^{13}C$ and C/N ratio values suggested a higher contribution of mangrove-derived organic matter in the inner forest, most probably because of its age, the high productivity of the stand, and the distance from the tidal creek that limits leaf litter flushing. At depth, beneath the mature *Rhizophora* stand and the transitional forest, an increased organic content combined with depleted $\delta^{13}C$ values and C/N ratio increase suggested an elevated contribution of vascular plant debris to the soil organic matter pool. We

suggest that this enrichment reflected the past mangrove forest before its destruction during the war.

4. Soil carbon stocks in the mature *Rhizophora* forest, down to one meter, represented almost three times the stock in the above-ground biomass. However, when considering only the upper soil, which was related to current forest development as evidenced by $\delta^{13}C$ values and C/N ratios, stocks in the soil and in the above-ground biomass were similar, and the carbon burial rate was lower than 1 Mg C ha^{-1} yr^{-1}.

In a future research effort, the net ecosystem productivity of the Can Gio mangrove forest, Southern Vietnam's largest, should be studied, possibly using the eddy-covariance technique. The influence of the monsoon on its productivity should also be considered.

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References

Adame, M.F., Kauffman, J.B., Medina, I., Gamboa, J.N., Torres, O., Caamal, J.P., Reza, M., Herrera-Silveira, J.A., 2013. Carbon Stocks of Tropical Coastal Wetlands within the

Karstic Landscape of the Mexican Caribbean. PLoS ONE 8, e56569. https://doi.org/10.1371/journal.pone.0056569

Adame, M.F., Santini, N.S., Tovilla, C., Vázquez-Lule, A., Castro, L., Guevara, M., 2015. Carbon stocks and soil sequestration rates of tropical riverine wetlands. Biogeosciences 12, 3805–3818. https://doi.org/10.5194/bg-12-3805-2015

Albers, T., 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. Wetlands ecology and management 23, 991–1004. https://doi.org/10.1007/s11273-015-9441-3

Alongi, D.M., 2014. Carbon Cycling and Storage in Mangrove Forests. Annual Review of Marine Science 6, 195–219. https://doi.org/10.1146/annurev-marine-010213-135020

Alongi, D.M., 2012. Carbon sequestration in mangrove forests. Carbon Management 3, 313–322. https://doi.org/10.4155/cmt.12.20

Alongi, D.M., 2008. Mangrove forests: Resilience, protection from tsunamis, and responses to global climate change. Estuarine, Coastal and Shelf Science 76, 1–13. https://doi.org/10.1016/j.ecss.2007.08.024

Alongi, D.M., Tirendi, F., Clough, B.F., 2000. Below-ground decomposition of organic matter in forests of the mangroves Rhizophora stylosa and Avicennia marina along the arid coast of Western Australia. Aquatic Botany 68, 97–122. https://doi.org/10.1016/S0304-3770(00)00110-8

Andersen, F., Kristensen, E., 1988. Oxygen microgradients in the rhizosphere of the mangrove Avicennia marina. Marine Ecology Progress Series 44, 201–204. https://doi.org/10.3354/meps044201

Balk, M., Keuskamp, J.A., Laanbroek, H.J., 2016. Potential for Sulfate Reduction in Mangrove Forest Soils: Comparison between Two Dominant Species of the Americas. Front. Microbiol. 7. https://doi.org/10.3389/fmicb.2016.01855

Balke, T., Bouma, T.J., Horstman, E.M., Webb, E.L., Erftemeijer, P.L., Herman, P.M., 2011. Windows of opportunity: thresholds to mangrove seedling establishment on tidal flats. Marine Ecology Progress Series 440, 1–9.

Banerjee, L.K., 1993. Influence of salinity on mangrove zonation, in: Towards the Rational Use of High Salinity Tolerant Plants, Tasks for Vegetation Science. Springer, Dordrecht, pp. 181–186. https://doi.org/10.1007/978-94-011-1858-3_19

Barbier, E.B., 2006. Natural barriers to natural disasters: replanting mangroves after the tsunami. Frontiers in Ecology and the Environment 4, 124–131. https://doi.org/10.1890/1540-9295(2006)004[0124:NBTNDR]2.0.CO;2

Bosire, J.O., Dahdouh-Guebas, F., Kairo, J.G., Kazungu, J., Dehairs, F., Koedam, N., 2005. Litter degradation and CN dynamics in reforested mangrove plantations at Gazi Bay, Kenya. Biological Conservation 126, 287–295. https://doi.org/10.1016/j.biocon.2005.06.007

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., others, 2008. Mangrove production and carbon sinks: a revision of global budget estimates. Global Biogeochemical Cycles 22. https://doi.org/10.1029/2007GB003052

Bouillon, S., Dahdouh-Guebas, F., Rao, A., Koedam, N., Dehairs, F., 2003. Sources of organic carbon in mangrove sediments: variability and possible ecological implications. Hydrobiologia 495, 33–39. https://doi.org/10.1023/A:1025411506526

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: Mangrove organic carbon burial rates. Global Biogeochemical Cycles 26. https://doi.org/10.1029/2012GB004375

Brunt, M.A., Davies, J.E. (Eds.), 1994. The Cayman Islands: Natural History and Biogeography, Monographiae Biologicae. Springer Netherlands.

Castillo, J.A.A., Apan, A.A., Maraseni, T.N., Salmo, S.G., 2017. Soil C quantities of mangrove forests, their competing land uses, and their spatial distribution in the coast of Honda Bay, Philippines. Geoderma 293, 82–90. https://doi.org/10.1016/j.geoderma.2017.01.025

Clough, B., 2014. Site Assessment Guidelines for Mangrove Rehabilitation in Bac Lieu Province, Vietnam 41.

Curran, M., Cole, M., Allaway, W.G., 1986. Root Aeration and Respiration in Young Mangrove Plants (Avicennia marina (Forsk.) Vierh.). J Exp Bot 37, 1225–1233. https://doi.org/10.1093/jxb/37.8.1225

David, F., C. Marchand, P. Taillardat, N. Thanh Nho, and T. Meziane. (2018a). Nutritional composition of suspended particulate matter in a tropical mangrove creek during a tidal cycle (Can Gio, Vietnam). Estuarine Coastal and Shelf Sciences 200, 126-130. https://doi.org/10.1016/j.ecss.2017.10.017

David, F., Meziane, T., Tran-Thi, N.-T., Truong Van, V., Thanh-Nho, N., Taillardat, P., Marchand, C., (2018b). Carbon biogeochemistry and CO_2 emissions in a human impacted and mangrove dominated tropical estuary (Can Gio, Vietnam). Biogeochemistry 1–15. https://doi.org/10.1007/s10533-018-0444-z

David, F., C. Marchand, N. Thanh-Nho, P. Taillardat, and T. Meziane. (2019). Trophic relationships and basal resources utilization in the Can Gio Mangrove Biosphere Reserve (Southern Vietnam). Journal of Sea Research 145, 35–43. https://doi.org/10.1016/j.seares.2018.12.006

DelVecchia, A.G., Bruno, J.F., Benninger, L., Alperin, M., Banerjee, O., Morales, J. de D., 2014. Organic carbon inventories in natural and restored Ecuadorian mangrove forests. PeerJ 2, e388. https://doi.org/10.7717/peerj.388

Donato, D.C., Kauffman, J.B., Murdiyarso, D., Kurnianto, S., Stidham, M., Kanninen, M., 2011. Mangroves among the most carbon-rich forests in the tropics. Nature Geoscience 4, 293–297. https://doi.org/10.1038/ngeo1123

Dung, L.V., Tue, N.T., Nhuan, M.T., Omori, K., 2016. Carbon storage in a restored mangrove forest in Can Gio Mangrove Forest Park, Mekong Delta, Vietnam. Forest Ecology and Management 380, 31–40. https://doi.org/10.1016/j.foreco.2016.08.032

Ellison, J.C., 1999. Impacts of sediment burial on mangroves. Marine Pollution Bulletin 37, 420–426. https://doi.org/10.1016/S0025-326X(98)00122-2

English, S.S., Wilkinson, C.C., Baker, V.V., 1997. Survey manual for tropical marine resources. Australian Institute of Marine Science.

Fromard, F., Puig, H., Mougin, E., Marty, G., Betoulle, J.L., Cadamuro, L., 1998. Structure, above-ground biomass and dynamics of mangrove ecosystems: new data from French Guiana. Oecologia 115, 39–53. https://doi.org/10.1007/s004420050489

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 Ecology and Biogeography 20, 154–159. https://doi.org/10.1111/j.1466-8238.2010.00584.x

Hawkins, S., Robertson, S., Thu Thuy, P., Xuan To, P., McNally, R., Van Cuong, C., Dart, P., Xuan Phuong, P., Brown, S., Vu, N., 2010. Roots in the water: legal frameworks for mangrove PES in Vietnam.

Hesse, P.R., 1961. Some differences between the soils of *Rhizophora* and *Avicennia* mangrove swamps in Sierra Leone. Plant and Soil 14, 335–346. https://doi.org/10.1007/BF01666292

Hiraishi, T., Krug, T., Tanabe, K., Srivastava, N., Baasansuren, J., Fukuda, M., Troxler, T.G., 2014. 2013 supplement to the 2006 IPCC guidelines for national greenhouse gas inventories: Wetlands. IPCC, Switzerland.

Hong, P.N., San, H.T., 1993. Mangroves of Vietnam. Iucn.

Hossain, M., 2014. Carbon pools and fluxes in Bruguiera parviflora dominated naturally growing mangrove forest of Peninsular Malaysia. Wetlands Ecol Manage 22, 15–23. https://doi.org/10.1007/s11273-013-9318-2.

Jacotot, A., Marchand, C., Rosenheim, B. E., Domack E., Allenbach, M., 2018. Mangrove soil carbon stocks along an elevation gradient: influence of the late Holocene marine regression (New Caledonia). Marine Geology 404, 60670.

Jennerjahn, T.C., Ittekkot, V., 1997. Organic matter in sediments in the mangrove areas and adjacent continental margins of Brazil. 1. Amino acids and hexosamines. Oceanolica Acta 20, 359–369.

Kathiresan, K., 2003. How do mangrove forests induce sedimentation? Revista de Biología Tropical 51, 355–360.

Kauffman, J.B., Donato, D., 2012. Protocols for the measurement, monitoring and reporting of structure, biomass and carbon stocks in mangrove forests. Center for International Forestry Research (CIFOR), Bogor, Indonesia.

Kauffman, J.B., Heider, C., Cole, T.G., Dwire, K.A., Donato, D.C., 2011. Ecosystem Carbon Stocks of Micronesian Mangrove Forests. Wetlands 31, 343–352. https://doi.org/10.1007/s13157-011-0148-9

Kendall, C.G.S.C., Skipwith, P.A. d. E., 1969. Geomorphology of a Recent Shallow-Water Carbonate Province: Khor Al Bazam, Trucial Coast, Southwest Persian Gulf. GSA Bulletin 80, 865–892. https://doi.org/10.1130/0016-7606(1969)80[865:GOARSC]2.0.CO;2

Khan, M.A., Aziz, I., 2001. Salinity tolerance in some mangrove species from Pakistan. Wetlands Ecology and Management 9, 229–233. https://doi.org/10.1023/A:1011112908069

Komiyama, A., Ong, J.E., Poungparn, S., 2008. Allometry, biomass, and productivity of mangrove forests: A review. Aquatic Botany 89, 128–137. https://doi.org/10.1016/j.aquabot.2007.12.006

Komiyama, A., Poungparn, S., Kato, S., 2005. Common allometric equations for estimating the tree weight of mangroves. Journal of Tropical Ecology 21, 471–477. https://doi.org/10.1017/S0266467405002476

Kristensen, E., Bouillon, S., Dittmar, T., Marchand, C., 2008. Organic carbon dynamics in mangrove ecosystems: A review. Aquatic Botany, Mangrove Ecology – Applications in Forestry and Costal Zone Management 89, 201–219. https://doi.org/10.1016/j.aquabot.2007.12.005

Lacerda, L.D., Ittekkot, V., Patchineelam, S.R., 1995. Biogeochemistry of Mangrove Soil Organic Matter: a Comparison BetweenRhizophoraandAvicenniaSoils in South-eastern Brazil. Estuarine, Coastal and Shelf Science 40, 713–720. https://doi.org/10.1006/ecss.1995.0048

Lal, R., 2005. Forest soils and carbon sequestration. Forest Ecology and Management, Forest Soils Research: Theory, Reality and its Role in TechnologY 220, 242–258. https://doi.org/10.1016/j.foreco.2005.08.015

Lamb, A.L., Wilson, G.P., Leng, M.J., 2006. A review of coastal palaeoclimate and relative sea-level reconstructions using δ13C and C/N ratios in organic material. Earth-Science Reviews 75, 29–57. https://doi.org/10.1016/j.earscirev.2005.10.003

Leopold, A., Marchand, C., Deborde, J., Chaduteau, C., Allenbach, M., 2013. Influence of mangrove zonation on CO 2 fluxes at the sediment–air interface (New Caledonia). Geoderma 202, 62–70. https://doi.org/10.1016/j.geoderma.2013.03.008

Liu, H., Ren, H., Hui, D., Wang, W., Liao, B., Cao, Q., 2014. Carbon stocks and potential carbon storage in the mangrove forests of China. Journal of Environmental Management 133, 86–93. https://doi.org/10.1016/j.jenvman.2013.11.037

Lunstrum, A., Chen, L., 2014. Soil carbon stocks and accumulation in young mangrove forests. Soil Biology and Biochemistry 75, 223–232. https://doi.org/10.1016/j.soilbio.2014.04.008

Luong, N.V., Tateishi, R., Hoan, N.T., 2015. Analysis of an impact of succession in mangrove forest association using remote sensing and GIS technology. Journal of Geography and Geology 7, 106. https://doi.org/10.5539/jgg.v7n1p106

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. Wetlands ecology and management 24, 245–261. https://doi.org/10.1007/s11273-016-9481-3

Marchand, C., 2017. Soil carbon stocks and burial rates along a mangrove forest chronosequence (French Guiana). Forest Ecology and Management 384, 92–99. https://doi.org/10.1016/j.foreco.2016.10.030

Marchand, C., Allenbach, M., Lallier-Vergès, E., 2011. Relationships between heavy metals distribution and organic matter cycling in mangrove sediments (Conception Bay, New Caledonia). Geoderma 160, 444–456. https://doi.org/10.1016/j.geoderma.2010.10.015

Marchand, C., Baltzer, F., Lallier-Vergès, E., Albéric, P., 2004. Pore-water chemistry in mangrove sediments: relationship with species composition and developmental stages (French Guiana). Marine Geology 208, 361–381. https://doi.org/10.1016/j.margeo.2004.04.015

Marchand, C., Disnar, J.-R., Lallier-Vergès, E., Lottier, N., 2005. Early diagenesis of carbohydrates and lignin in mangrove sediments subject to variable redox conditions (French Guiana). Geochimica et Cosmochimica Acta 69, 131–142. https://doi.org/10.1016/j.gca.2004.06.016

Marchand, C., Fernandez, J.-M., Moreton, B., Landi, L., Lallier-Vergès, E., Baltzer, F., 2012. The partitioning of transitional metals (Fe, Mn, Ni, Cr) in mangrove sediments downstream of a ferralitized ultramafic watershed (New Caledonia). Chemical Geology 300–301, 70–80. https://doi.org/10.1016/j.chemgeo.2012.01.018

McKee, K.L., 1993. Soil physicochemical patterns and mangrove species distribution–reciprocal effects. Journal of ecology 477–487. http://dx.doi.org/10.2307/2261526

McKee, K.L., Mendelssohn, I.A., 1987. Root metabolism in the black mangrove (Avicennia germinans (L.) L): Response to hypoxia. Environmental and Experimental Botany 27, 147–156. https://doi.org/10.1016/0098-8472(87)90065-7

Mizanur Rahman, M., Nabiul Islam Khan, M., Fazlul Hoque, A.K., Ahmed, I., 2015. Carbon stock in the Sundarbans mangrove forest: spatial variations in vegetation types and salinity zones. Wetlands Ecology and Management 23, 269–283. https://doi.org/10.1007/s11273-014-9379-x

Mukherjee, N., Sutherland, W.J., Dicks, L., Hugé, J., Koedam, N., Dahdouh-Guebas, F., 2014. Ecosystem Service Valuations of Mangrove Ecosystems to Inform Decision Making and Future Valuation Exercises. PLoS One 9. https://doi.org/10.1371/journal.pone.0107706

Mumby, P.J., Edwards, A.J., Arias-González, J.E., Lindeman, K.C., Blackwell, P.G., Gall, A., Gorczynska, M.I., Harborne, A.R., Pescod, C.L., Renken, H., 2004. Mangroves enhance the biomass of coral reef fish communities in the Caribbean. Nature 427, 533–536. https://doi.org/10.1038/nature02286

Nagelkerken, I., Blaber, S.J.M., Bouillon, S., Green, P., Haywood, M., Kirton, L.G., Meynecke, J.-O., Pawlik, J., Penrose, H.M., Sasekumar, A., Somerfield, P.J., 2008. The habitat function of mangroves for terrestrial and marine fauna: A review. Aquatic Botany, Mangrove Ecology – Applications in Forestry and Costal Zone Management 89, 155–185. https://doi.org/10.1016/j.aquabot.2007.12.007

Naidoo, G., Naidoo, K., 2017. Are pioneer mangroves more vulnerable to oil pollution than later successional species? Mar. Pollut. Bull. 121, 135–142. https://doi.org/10.1016/j.marpolbul.2017.05.067

Nam, V.N., Sasmito, S.D., Murdiyarso, D., Purbopuspito, J., MacKenzie, R.A., 2016. Carbon stocks in artificially and naturally regenerated mangrove ecosystems in the Mekong Delta. Wetlands ecology and management 24, 231–244. https://doi.org/10.1007/s11273-015-9479-2

Noël, V., Juillot, F., Morin, G., Marchand, C., Ona-Nguema, G., Viollier, E., Prévot, F., Dublet, G., Maillot, F., Delbes, L., Marakovic, G., Bargar, J.R., Brown, G.E., 2017. Oxidation of Ni-Rich Mangrove Sediments after Isolation from the Sea (Dumbea Bay, New Caledonia): Fe and Ni Behavior and Environmental Implications. ACS Earth and Space Chemistry 1, 455–464. https://doi.org/10.1021/acsearthspacechem.7b00005

Noël, V., Marchand, C., Juillot, F., Ona-Nguema, G., Viollier, E., Marakovic, G., Olivi, L., Delbes, L., Gelebart, F., Morin, G., 2014. EXAFS analysis of iron cycling in mangrove sediments downstream a lateritized ultramafic watershed (Vavouto Bay, New Caledonia). Geochimica et Cosmochimica Acta 136, 211–228. https://doi.org/10.1016/j.gca.2014.03.019

Pissierssens, P., 2002. Manual on Sea Level Measurement and Interpretation. UNESCO.

Prasad, M.B.K., Kumar, A., Ramanathan, A.L., Datta, D.K., 2017. Sources and dynamics of sedimentary organic matter in Sundarban mangrove estuary from Indo-Gangetic delta. Ecological Processes 6. https://doi.org/10.1186/s13717-017-0076-6

Proisy, C., Gratiot, N., Anthony, E.J., Gardel, A., Fromard, F., Heuret, P., 2009. Mud bank colonization by opportunistic mangroves: A case study from French Guiana using lidar data. Continental Shelf Research, On the dynamics of mud deposits in coastal areas 29, 632–641. https://doi.org/10.1016/j.csr.2008.09.017

Putz, F.E., Chan, H.T., 1986. Tree growth, dynamics, and productivity in a mature mangrove forest in Malaysia. Forest ecology and management 17, 211–230. https://doi.org/10.1016/0378-1127(86)90113-1

Rivera-Monroy, V.H., Torres, L.A., Bahamon, N., Newmark, F., Twilley, R.R., 1999. The Potential Use of Mangrove Forests as Nitrogen Sinks of Shrimp Aquaculture Pond Effluents: The Role of Denitrification. Journal of the World Aquaculture Society 30, 12–25. https://doi.org/10.1111/j.1749-7345.1999.tb00313.x

Ross, P., 1975. The mangroves of South Vietnam: the impact of military use of herbicides, in: Proceedings of the International Symposium on Biology and Management of Mangroves. Gainesville, Florida: Institute of Food and Agricultural Sciences, University of Florida. pp. 695–709.

Saenger, P., 2002. Introduction: The Mangrove Environment, in: Mangrove Ecology, Silviculture and Conservation. Springer, pp. 1–10.

Saenger, P., Snedaker, S.C., 1993. Pantropical trends in mangrove above-ground biomass and annual litterfall. Oecologia 96, 293–299. https://doi.org/10.1007/BF00317496

Sanders, C.J., Smoak, J.M., Naidu, A.S., Sanders, L.M., Patchineelam, S.R., 2010. Organic carbon burial in a mangrove forest, margin and intertidal mudflat. Estuarine, Coastal and Shelf Science 90, 168–172. https://doi.org/10.1016/j.ecss.2010.08.013

Schmitt, K., Duke, N.C., 2016. Mangrove Management, Assessment, and Monitoring, in: Tropical Forestry Handbook. Springer, pp. 1725–1759.

Thanh-Nho, N., Strady, E., Nhu-Trang, T.-T., David, F., Marchand, C., 2018. Trace metals partitioning between particulate and dissolved phases along a tropical mangrove estuary (Can Gio, Vietnam). Chemosphere 196, 311–322. https://doi.org/10.1016/j.chemosphere.2017.12.189

Thibodeau, F.R., Nickerson, N.H., 1986. Differential Oxidation of Mangrove Substrate by Avicennia germinans and Rhizophora mangle. American Journal of Botany 73, 512–516. https://doi.org/10.2307/2444255

Tue, N.T., Dung, L.V., Nhuan, M.T., Omori, K., 2014. Carbon storage of a tropical mangrove forest in Mui Ca Mau National Park, Vietnam. Catena 121, 119–126. https://doi.org/10.1016/j.catena.2014.05.008

Ukpong, I.E., 1997. Vegetation and its relation to soil nutrient and salinity in the Calabar mangrove swamp, Nigeria. Mangroves and Salt Marshes 1, 211–218. https://doi.org/10.1023/A:1009952700317

Ukpong, L.E., 1995. Vegetation and soil acidity of a mangrove swamp in southeastern Nigeria. Soil use and management 11, 141–144. https://doi.org/10.1111/j.1475-2743.1995.tb00512.x

Vafaei, S., Soosani, J., Adeli, K., Fadaei, H., Naghavi, H., Pham, T.D., Tien Bui, D., 2018. Improving Accuracy Estimation of Forest Aboveground Biomass Based on Incorporation of ALOS-2 PALSAR-2 and Sentinel-2A Imagery and Machine Learning: A Case Study of the Hyrcanian Forest Area (Iran). Remote Sensing 10, 172. https://doi.org/10.3390/rs10020172

Vinh, T.V., Marchand C., Linh, T.V.K., Vinh, D.D., Allenbach, M. 2019. Allometric models to estimate above-ground biomass and carbon stocks in a Rhizophora apiculata tropical mangrove forest (Southern Vietnam). Forest Ecology and Management. https://doi.org/10.1016/j.foreco.2018.12.017

Vinh, T.V., Allenbach, M., Linh, T.V.K., Marchand C., 2020. Changes in leaf litter quality during its decomposition in a tropical planted mangrove forest (Can Gio, Vietnam). Frontiers in Environmental Science. Biogeochemical Dynamics. https://doi: 10.3389/fenvs.2020.00010

Vo, Q.T., Oppelt, N., Leinenkugel, P., Kuenzer, C., 2013. Remote sensing in mapping mangrove ecosystems—An object-based approach. Remote Sensing 5, 183–201. https://doi:10.3390/rs501018

Wachid, M.N., Hapsara, R.P., Cahyo, R.D., Wahyu, G.N., Syarif, A.M., Umarhadi, D.A., Fitriani, A.N., Ramadhanningrum, D.P., Widyatmanti, W., 2017. Mangrove canopy density analysis using Sentinel-2A imagery satellite data, in: IOP Conference Series: Earth and Environmental Science. IOP Publishing, p. 012020.

Walsh, G.E., 1974. Mangroves: a review, in: Ecology of Halophytes. Elsevier, pp. 51–174. https://doi.org/10.1016/B978-0-12-586450-3.X5001-X

Wang, G., Guan, D., Peart, M.R., Chen, Y., Peng, Y., 2013. Ecosystem carbon stocks of mangrove forest in Yingluo Bay, Guangdong Province of South China. Forest Ecology and Management 310, 539–546. https://doi.org/10.1016/j.foreco.2013.08.045

Table 1. Vegetation structure in the different zones. MF: Mudflat; FF: Fringe forest; TF: Transition forest; IF: Interior forest; Aa: *Avicennia alba*; Ao: *Avicenia officinalis*; Ct: *Ceriops tagal*; Aa: *Sonneratia alba*; Ra: *Rhizophora apiculata*, AGC above-ground C stock, SC soil C stock.

Sites	Species	DBH (cm)	Tree density ha^{-1}	Basal area (m^2 ha^{-1})
MF	-	-	-	-
FF	*Aa, Ao*	10.7	1327	11.9
TF	*Aa, Ao, Ct, Sa, Ra*	8.5	3727	21.2
IF	*Ra*	18.6	1127	28.4

Table 2: Two-way ANOVA tests for pH, Eh and salinity values showing the effect of sampling site and seasons on soil parameters. * and *** indicate statistically significant effects for p-Values < 0.05 and < 0.001, respectively .ns means no statistically significant difference ($p > 0.05$).

Parameters	n	Source of variation		
		Sites	Seasons	Interaction
pH	150	48.9***	5.9*	22.8***
Eh	150	19.1***	1.4 ns	11.9***
Pore-water salinity	150	1137.7***	203.4***	23.7***

Figures captions

Figure 1: Study sites location. (A) Vietnam map, (B) Can Gio Estuary and (C) Site study with three transects from the mudflat to the inner forest.

Figure 2: Mangrove distribution along the transect at Can Gio mangrove, (a) and (b) distribution of mangrove species from river towards the land, (c) elevation of the intertidal zone and mangrove zonation.

Figure 3: Mean salinity, Eh and pH values beneath the different mangrove zones studied: Mudflat (a, b, c), fringe forest (d, e, f), transitional forest (g, h, k), and inner forest (l, m, n). Orange lines present the values measured during the dry season and blue lines present the values measured during the rainy season.

Figure 4: C contents (a), C/N ratios (b) and δ^{13}C values (c) profiles. Black, light brown, orange, blue, green lines represent mudflat, fringe forest, transition forest, and mature *Rhizophora,* (d) picture of buried dead trunk of *Rhizophora* below the actual root system.

Figure 5: Ecosystem C stocks in the different sites along the elevation gradient. Green bars represent above-ground C stock, grey bars represent soil C stocks from 0 to 40 cm depth, orange bars represent below-ground C stocks from 40 to 100 cm depth, blue bars represent below-ground C stock from 0 to 100 cm depth. Solid vertical black lines present the standard deviation (SD).