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MANGROVES IN CHANGING ENVIRONMENTS

# Growth performance and structure of a mangrove afforestation project on a former seagrass bed, Mindanao Island, Philippines

**Sahadev Sharma · Kazuo Nadaoka · Masahiro Nakaoka · Wilfredo H. Uy · Richard A. MacKenzie · Daniel A. Friess · Miguel D. Fortes**

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**Abstract** The Philippines has lost nearly 70% of its natural mangrove cover since the early 1900s. As a result, large investments have been made to restore mangrove forests and the many ecosystem services that they provide. Most of these restoration efforts have been through outplanting of *Rhizophora* sp. seedlings, many of which have failed because the proper hydrological and ecological conditions were not properly assessed. Other afforestation projects involved planting seedlings in inappropriate places (e.g., seagrass beds, mudflats) that resulted in replacing one valuable ecosystem with another. The aim of this research was to investigate the growth of 3-, 9-, and 21-year-old afforested stands of *Rhizophora* spp. mangrove forest. We also investigated the impact of these plantations on local seagrass beds. The total aboveground biomass was 42.6, 74.4, and 111.7 Mg ha$^{-1}$ for the 3-, 9-, and 21-year-old mangrove stands, respectively. Seagrass bed cover decreased under the closed canopy of the mangrove due to reduced photosynthetically active radiation and competition for growing space. This study shows that mangroves can grow to some extent on seagrass beds, though mangrove planting in these areas could eventually lead to seagrass loss. Thus, mangroves should not be planted in areas that are naturally occupied by other ecologically important ecosystems. The purpose of mangrove restoration should be clear and efforts should be focused on formerly deforested or degraded

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S. Sharma · K. Nadaoka
Department of Mechanical and Environmental Informatics, Tokyo Institute of Technology, Tokyo, Japan

S. Sharma
Department of Natural Resources and Environmental Management, University of Hawai'I at Manoa, Honolulu, HI, USA

M. Nakaoka
Akkeshi Marine Station, Field Science Center for Northern Biosphere, Hokkaido University, Akkeshi, Hokkaido 088-1113, Japan

W. H. Uy
Institute of Fisheries Research and Development, Mindanao State University at Naawan, Naawan, Misamis Oriental, Philippines

S. Sharma (✉) · R. A. MacKenzie
USDA Forest Service, Pacific Southwest Research Station, Institute of Pacific Islands Forestry, 60 Nowelo Street, Hilo, HI 96720, USA
e-mail: ssharma@hawaii.edu

areas. Additional studies are needed from different locations to understand how mangrove planting in seagrass beds impacts growth performance and ecological functions of the latter ecosystem.

**Keywords** Carbon sequestration · Blue carbon · Leaf area index · Carbon emission · Deforestation · Restoration

## Introduction

Mangrove forests are highly productive ecosystems that provide many ecosystem services to surrounding coastal areas (Oudenhoven et al., 2015). These ecosystem services have been valued at $4185 USD per hectare per year in Southeast Asia (Brander et al., 2012). Despite their value, large areas of mangroves have been cleared for aquaculture ponds, cut for firewood and timber, or cleared for residential and urban development (Dewalt et al., 1996; Uddin et al., 2014). The Philippines experienced particularly large rates of mangrove deforestation in the 20th century, with an estimated loss of nearly half of their natural mangrove forests (Baconguis et al., 1990; Primavera, 1995). Deforestation rates from the beginning of the 21st century in the Philippines are much lower at 0.04% per year, with 37% of losses due to the expansion of aquaculture and to an extent, oil palm (Richards & Friess, 2016, see also Hamilton & Casey, 2016). Future threats include climate change as Philippines' mangroves are vulnerable to accelerated sea-level rise and changing rainfall patterns (Lovelock et al., 2015; Ward et al., 2016).

The loss of mangroves in the Philippines coupled with their recent recognized value at providing ecosystem services to coastal communities has resulted in an increase in conservation and restoration of these forested ecosystems across the Philippines. However, several conservation and management challenges exist in the Philippines that are often related to overlapping bureaucracies (Friess et al., 2016). As a result, conservation of mangrove forests through protection is often unsuccessful and deforestation continues. Alternatively, the rehabilitation of degraded mangroves or the afforestation of mangroves in new areas to increase carbon stocks or provide protection from storms has become a much more popular approach. For example, the Philippines has invested \$100–500 USD ha$^{-1}$ into mangrove restoration programs over the last decade (Primavera & Esteban, 2008) and most recently the Department of Environment and Natural Resources (DENR) allocated $22 million USD for mangrove and beach forest rehabilitation projects after super typhoon Haiyan devastated the Eastern Visayas region in 2013.

Unfortunately, an estimated 70–80% of these mangrove restoration projects have failed because they involved the outplanting of inappropriate species (e.g., *Rhizophora* spp.) in unsuitable hydrological or geomorphological settings (Lewis, 2005; Primavera & Esteban, 2008; Samson & Rollen, 2008). Many of these projects also ignored key hydrological thresholds that determine mangrove establishment and long-term stability (Friess et al., 2012). Afforestation has resulted in the replacement other valuable ecosystems (i.e., seagrass beds) with mangrove forests. This is due to rehabilitation planners often choosing uncontested areas such as seagrass beds that are not former aquaculture ponds produced by illegal encroachment and still claimed by encroachers (Primavera & Esteban, 2008). These areas are also easier to plant in due to the lack of complex land tenure arrangements found in more landward coastal areas and because they provide large flat areas that are ideal for large-scale afforestation. The replacement of seagrass beds by mangrove plantations are expected to have negative repercussion as they also provide numerous ecological services to the surrounding coastal environment (Wright & Jones, 2006; Fourqurean et al., 2012; Nakaoka et al., 2014). Furthermore, it is unclear if mangrove plantations even provide the same level of ecosystem services or benefits as the intact ones they are replacing.

Due to sub-optimal hydrological conditions for mangrove growth, we hypothesized that mangrove afforestation on seagrass beds would lead to reduced mangrove growth and structure compared to other reforested mangrove stands. This study aimed to do the

D. A. Friess
Department of Geography, National University of Singapore, 1 Arts Link, Singapore 117570, Singapore

M. D. Fortes
The Marine Science Institute, CS, University of the Philippines, Diliman, 1101 Quezon City, Philippines

following: (1) to assess the forest structure and development of aboveground biomass (ABG) for fringing planted *Rhizophora* spp. stands that vary in ages of 3, 9, and 21 years old; (2) to investigate the relationships between ABG and structural properties of the planted stand; and (3) to examine the effects of mangrove structural characteristics on seagrass bed cover.

## Materials and methods

### Description of the study area

We estimated stand attributes, leaf area index, aboveground biomass, and productivity of mangroves as well as seagrass bed cover under mangrove canopy in 3-, 9-, and 21-year-old afforested mangrove stands in Laguindingan, Misamis Oriental, Philippines (Lat 8.623944°; Long 124.465557°) (Fig. 1). The age of the mangroves were based on personal communication with the agriculture office and local communities. The forest was planted on intertidal seagrass beds by the local community with assistance from the DENR, in order to protect the coastal village from typhoons, as typhoons in this location are common (Honda et al., 2013). Tidal range in seagrass beds during low and high tides was 0.5–1.0 and 1.5–2.0 m, respectively, while in mangrove areas 0.5–1.0 m at low tide and 1.0–1.5 m at high tide (Honda et al., 2013).

Mangroves at the study site are surrounded by coral reefs, and are growing over former reef flats, seagrass beds, and coral sands fringing the island. The site is dominated by *Rhizophora mucronata* Lam. and *R. apiculata* Blume., with naturally recruited *Avicennia lanata* Ridley and *Sonneratia alba* Sm. also present. Dominant seagrass species at the study site were *Thalassia hemprichii* (Ehrenb. ex Solms) Asch. (63.6%) and *Enhalus acoroides* (L.f.) Royle (4.3%) (Honda et al., 2013). Sea surface water temperature at the study site was 30.4°C in September and 28.8°C in March (Honda et al., 2013). Average soil salinity measured during this study was 36.0 ± 3.5‰, pH was 7.07 ± 0.09, and oxidation redox potential was −235.40 ± 63.13 mV.

### Study design, estimation of aboveground biomass, and seagrass area

Fieldwork was conducted in September 2013. Four belt-transects perpendicular to the mangrove-water (or seagrass) interface were established across the entire width of the planted mangrove forest to cover all age groups (Fig. 1). This sample design was chosen in

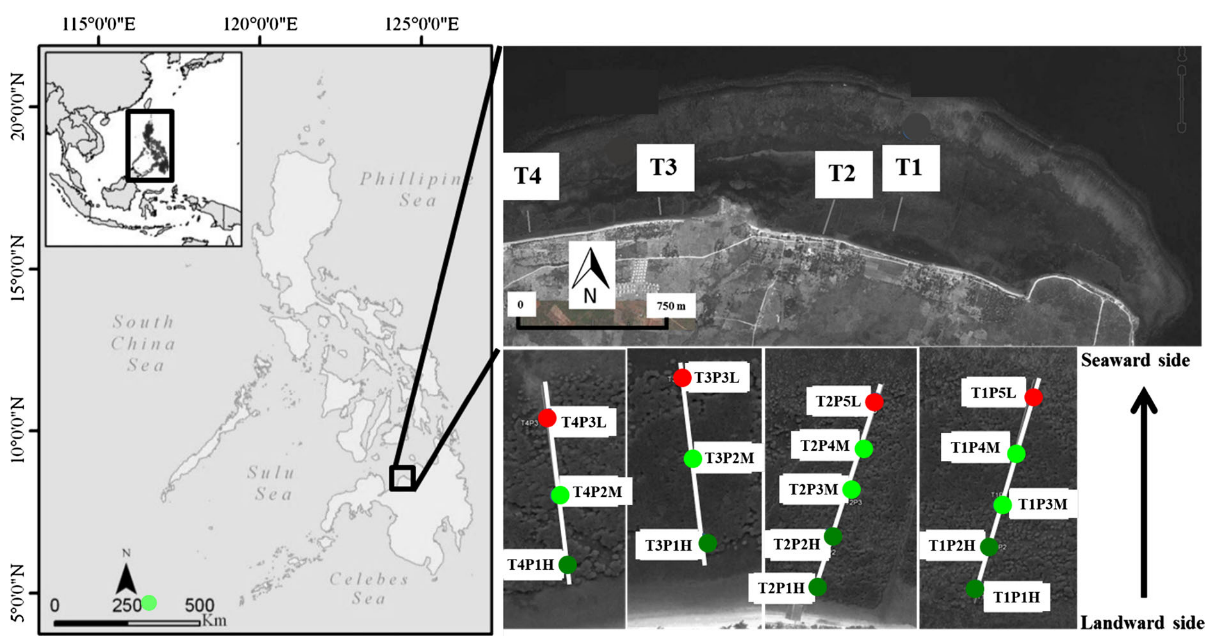


**Fig. 1** Location of the study area (**a**), location of transects (*T1*, *T2*, *T3*, and *T4*) locations (**b**), and location of plots from landward to seaward side in each transects. *Dark green color* plots are productive, *light green color* plots and medium productive and *red color* plots are less productive. *H* landward plots, *M* mid seaward plots, *L* lower seaward plots

order to representatively sample mangrove characteristics over a tidal inundation gradient. Two of the transects were 170-m long (transect 1 and 2) and two were 92-m long (transect 3 and 4). Transects were composed of 7-m radius (0.0154 ha) circular plots established at 25 m intervals (sensu Kauffman & Donato, 2012), with 16 plots measured in total. Forest structure and composition were then sampled from within each plot along each belt transect from the upland to the seaward edge. All trees >1.3 m tree height were numbered, counted, and the diameter at breast height (1.30 m, dbh) measured within each plot. For *Rhizophora* spp., dbh was measured 30 cm above the highest prop root.

Aboveground tree biomass was determined using species-specific allometric equations with dbh as the independent variable. The aboveground biomass (ABG) was computed using the following allometric equation for *Rhizophora* spp.:

$$W = 0.235\ \mathrm{dbh}^{2.42}, \tag{1}$$

where W is the ABG (in kg) and dbh is the diameter at 30 cm above the highest prop root (in cm) (Ong et al., 2004). We used this equation due to similar climatic conditions for the Philippines and Malaysia. Salmo et al. (2013) also used this equation to calculate biomass of *Rhizophora* spp. mangroves in the Philippines.

We used guidelines for the rapid assessment of seagrass habitats to estimate percentage cover of seagrass (McKenzie et al., 2003). A 0.25 $m^2$ gridded quadrat was haphazardly thrown three times within the 7-m radius plot and percent area of seagrass within the quadrat was then visually estimated and averaged across the three observations.

### Estimation of leaf area index

Within all plots, leaf area index was estimated using the light interception method (Clough et al., 1997). The light intensity measurement was performed at 10:00 a.m. to 14:00 p.m. on overcast days in September 2013. A total of 56 readings were taken for each plot using a data logger (LI-1400; LI-COR, USA) and a pair of horizontally placed quantum sensors (LI-190SA; LI-COR, USA). One was mounted 2 m above the canopy, while the other was mounted on a measuring rod and extended into the canopy at each plot. Light intensity relative to that above the canopy (relative light intensity) for all measured points was calculated on the basis of the stationary sensor placed above the canopy. Measurements of light absorption by the forest canopy are used to estimate leaf area index, $L$ ($m^2$ leaf area $m^{-2}$ ground area) using the equation:

$$L = \frac{\log_e\left(\frac{I}{I_0}\right)}{-k} \times \cos\theta, \tag{2}$$

where $I$ and $I_0$ are the mean photosynthetically active radiation under the canopy and incident radiation, respectively; $k$ is the canopy light extinction coefficient (0.5); and $\theta$ is a solar zenith angle, which was used to correct the $L$ for the latitude of this forest.

### Data analyses

We calculated the mean and standard deviation (SD) of stand attributes, leaf area index, aboveground biomass, and seagrass bed cover under mangrove canopy from sixteen plots. Data normality was assessed using Shapiro–Wilk test. Pearson's correlation coefficient ($r$) analysis was used to determine the relationships among stand structural parameters and ABG. Pearson's correlation coefficient assesses the relationship between two variables, which can be described with a linear function. Growth rates were estimated by dividing stand structural parameter values by respective age of that stand.

## Results

### Stand structural attributes

At the study site, *R. mucronata* and *R. apiculata* were the dominant mangrove species present, which are very common species used for mangrove afforestation projects in the Philippines and Southeast Asia. The planted mangroves were characterized by high densities of small *Rhizophora* spp. trees of 4903, 4719, and 5617 $ha^{-1}$ in 3-, 9-, and 21-year-old stands, respectively (Table 1). In natural settings we would expect tree density to decrease with stand age, though in our study the 21-year stand had a higher tree density compared to 3- and 9-year-old stands (Table 1). The mean diameter of trees increased with age, from 4.0,

**Table 1** Structural characteristics, aboveground biomass (ABG), and seagrass cover in each plot at the study site

| Plot | Stand age (year) | Stem density ($ha^{-1}$) (no. of actual tree in a plot) | dbh (cm) | Height (m) | Basal area ($m^2$ $ha^{-1}$) | Leaf area index ($m^2$ $m^{-2}$) | ABG (Mg $ha^{-1}$) | Seagrass cover (%) |
|---|---|---|---|---|---|---|---|---|
| T1P1H | 21 | 6299 (97) | 5.19 | 5.51 | 14.54 | 3.13 | 92.36 | 5 |
| T1P2H | 21 | 3247 (50) | 4.21 | 4.19 | 5.04 | 1.98 | 29.61 | 0 |
| T1P3M | 9 | 6364 (98) | 5.29 | 5.64 | 15.27 | 3.27 | 97.47 | 0 |
| T1P4M | 9 | 6883 (106) | 5.02 | 5.29 | 15.02 | 3.42 | 94.52 | 15 |
| T1P5L | 3 | 6494 (100) | 4.49 | 4.60 | 11.02 | 1.64 | 64.99 | 35 |
| T2P1H | 21 | 3961 (61) | 6.62 | 7.08 | 14.25 | 3.25 | 100.23 | 10 |
| T2P2H | 21 | 3247 (50) | 6.73 | 7.16 | 12.95 | 3.73 | 91.99 | 40 |
| T2P3M | 9 | 3636 (56) | 6.13 | 6.69 | 11.53 | 2.98 | 77.54 | 30 |
| T2P4M | 9 | 5065 (78) | 4.80 | 4.98 | 10.11 | 2.49 | 62.33 | 50 |
| T2P5L | 3 | 3312 (51) | 3.51 | 3.09 | 3.63 | 1.67 | 20.23 | 60 |
| T3P1H | 21 | 6494 (100) | 5.99 | 6.52 | 19.82 | 3.10 | 133.23 | 0 |
| T3P2M | 21 | 8701 (134) | 5.52 | 6.01 | 24.89 | 3.99 | 159.77 | 0 |
| T3P3L | 9 | 3766 (58) | 5.41 | 5.90 | 9.46 | 1.94 | 59.62 | 50 |
| T4P1H | 21 | 7468 (115) | 6.02 | 6.51 | 23.91 | 4.13 | 160.62 | 0 |
| T4P2M | 21 | 5520 (85) | 6.30 | 6.89 | 18.45 | 2.75 | 125.46 | 25 |
| T4P3L | 9 | 2597 (40) | 6.13 | 6.70 | 8.21 | 1.98 | 55.13 | 50 |
| Mean $\pm$ SE ($n = 2$) | 3 | 4903 $\pm$ 1591 | 4.0 $\pm$ 0.5 | 3.8 $\pm$ 0.8 | 7.3 $\pm$ 3.7 | 1.7 $\pm$ 0.02 | 42.6 $\pm$ 22.4 | 47.5 $\pm$ 12.5 |
| Mean $\pm$ SE ($n = 6$) | 9 | 4719 $\pm$ 685 | 5.5 $\pm$ 0.2 | 5.9 $\pm$ 0.3 | 11.6 $\pm$ 1.2 | 2.7 $\pm$ 0.3 | 74.4 $\pm$ 7.5 | 32.5 $\pm$ 8.7 |
| Mean $\pm$ SE ($n = 8$) | 21 | 5617 $\pm$ 709 | 5.8 $\pm$ 0.4 | 6.2 $\pm$ 0.4 | 16.7 $\pm$ 2.3 | 3.3 $\pm$ 0.2 | 111.7 $\pm$ 15.2 | 10.0 $\pm$ 5.3 |

5.5 to 5.8 cm for 3-, 9-, and 21-year stands, respectively (Table 1). However, the annual diameter increment rate showed the opposite pattern, with the 21- and 9-year stands increasing at 0.3 and 0.6 cm $year^{-1}$, respectively, which is lower than the 1.3 cm $year^{-1}$ recorded for the 3-year-old stand in this study. This result showed that initial growth is higher than for old growth stands. Tree height for 3-, 9-, and 21-year-old stands was 3.8, 5.9, and 6.2 m, respectively (Table 1). Tree height and diameter showed a similar trend (Table 1). There was also a significant and strong correlation between tree diameter and height (Table 2). Basal area of the 21-year stands ranged from 5.04 to 24.89 $m^2$ $ha^{-1}$ with a mean ($\pm$SD) value of 16.7 $\pm$ 2.3 $m^2$ $ha^{-1}$ (Table 1). Most of the basal area is accounted for by trees in the 2.0–9.0 cm dbh size class range. Mean basal area values for 3- and 9-year stands were 7.3 and 11.6 $m^2$ $ha^{-1}$, respectively (Table 1).

Tree basal area was significantly and positively correlated to stem density and leaf area index (Table 2). Leaf area index (LAI) varied from 1.98 to 4.13 $m^2$ $m^{-2}$ with mean ($\pm$SD) value of 3.3 $\pm$ 0.2 $m^2$ $m^{-2}$ for 21-year-old stand (Table 1). LAI was 1.7 and 2.7 $m^2$ $m^{-2}$ for 3- and 9-year-old stands, respectively, which suggested that LAI increased with stand growth (Table 1).

## Aboveground biomass (ABG)

The mean ($\pm$SD) ABG was 111.7 $\pm$ 15.2 Mg $ha^{-1}$ with minimum and maximum values of 20.23 and 160.62 Mg $ha^{-1}$ for 21-year-old stands from the 4 transects (Table 1). All 21-year-old stands show higher ABG values except for plot T1P2H, where trees had been harvested by the local community. Mean ABG was lower for the 3- and 9-year stands, at 42.6 and 74.4 Mg $ha^{-1}$ (Table 1). Biomass accumulated at 5.3 Mg $ha^{-1}$ $year^{-1}$ for the 21-year-old stand, which was a lower rate compared to the 3- and 9-year-old stands. The majority of ABG was due to trees with a dbh $<$ 10 cm; landward trees with dbh $>$ 10 cm

**Table 2** Pearson's correlation coefficient (2-tailed) of the stand structure, ABG, and seagrass cover on *Rhizophora* species stand in Laguindingan ($n = 16$)

| | dbh (cm) | Height (m) | Stem density ($ha^{-1}$) | Basal area ($m^2\ ha^{-1}$) | Leaf area index ($m^2\ m^{-2}$) | Seagrass cover (%) |
|---|---|---|---|---|---|---|
| Height (m) | 0.99*** | | | | | |
| Stem density (n $ha^{-1}$) | −0.02 | −0.03 | | | | |
| Basal area ($m^2\ ha^{-1}$) | 0.53* | 0.58* | 0.81*** | | | |
| Leaf area index ($m^2\ m^{-2}$) | 0.23 | 0.27 | 0.81*** | 0.85*** | | |
| Seagrass cover (%) | −0.19 | −0.22 | −0.62** | −0.68** | −0.77** | |
| ABG (Mg $ha^{-1}$) | 0.60* | 0.62* | 0.75*** | 0.99*** | 0.82*** | −0.63** |

Levels of significances are shown as: *0.05, **0.01, and ***0.001

only contributed to 5–20% of total ABG in each transect (Fig. 2; Supplementary material Fig. 1S). ABG was positively and significantly correlated to stem density, basal area, and LAI (Table 2). The significant positive correlation between biomass and stem density as well as basal area suggests that stand structure is closely related to biomass (Table 2).

Seagrass bed cover

Seaward plots showed a higher percentage of seagrass cover under mangrove canopy compared to landward side and middle plots (Table 1). Landward plots showed lower seagrass cover because they are not permanently inundated, though seagrass needs to be submerged more than diurnally (Table 1; Fig. 1). Furthermore, plots T1P2H, T1P3 M, T3P1H, T3P2 M, and T4P1H had no seagrass cover due to high density of mangrove trees and closed canopy cover (Table 1). Seagrass area cover changed with stand age, with older mangrove stands having lower seagrass cover compared to younger mangrove stands (Table 1). Seagrass area cover under 3-, 9-, and 21-year stands was 47.5, 32.5, and 10.0%, respectively (Table 1).

## Discussion

Afforestation success and changes in mangrove forest structure with age

The mangrove afforestation project instigated by the local community at Laguindingan was somewhat successful, with *Rhizophora*-dominated stands becoming denser with stand age (Table 1). The 9-year-old stands in this study had similar tree densities as those recorded in an 8-year planted *R. mucronata* stand (4244 $ha^{-1}$) in Pangapisan, northern Philippines (Salmo et al., 2013). The average tree density from 3-year-old stands was lower (9623 $ha^{-1}$) than similar aged stands in Sawi Bay, Thailand (Table 3). While natural mangrove stands generally show a decrease in tree density with age (Khan et al., 2013; Goessens et al., 2014; Sillanpää et al., 2017), the seaward 3-year-old stands from our study had lower tree densities compared to higher aged mangrove stands. Several factors may have contributed to this pattern. First, the older and more landward mangrove stands were probably at more hydrologically appropriate locations. Mangroves show species distribution along tidal inundation gradients (Watson, 1928), with establishment and growth rates greater at elevations where tidal inundation and thus submergence is below species-specific thresholds of physiological tolerance (Ball, 1988; Friess et al., 2012). This is further supported by the fact that *Rhizophora* spp. seedlings planted in low elevation, seaward zones that are not optimal for this species exhibit high mortality and low growth rates. As a result, stand structure maturity of these plantations can be delayed by 10 years (Salmo & Juanico, 2015). Alternatively, more favorable physical conditions result in higher mangrove growth rates, stands that reach maturity at quicker rates, and the provision of new recruits to the original planted mangroves. In our study, the lower number of older *Rhizophora* spp. trees with a dbh > 10 cm may explain the lack of regeneration, as seedling production and establishment is associated with high adult stem density and distance to parent trees (Delgado

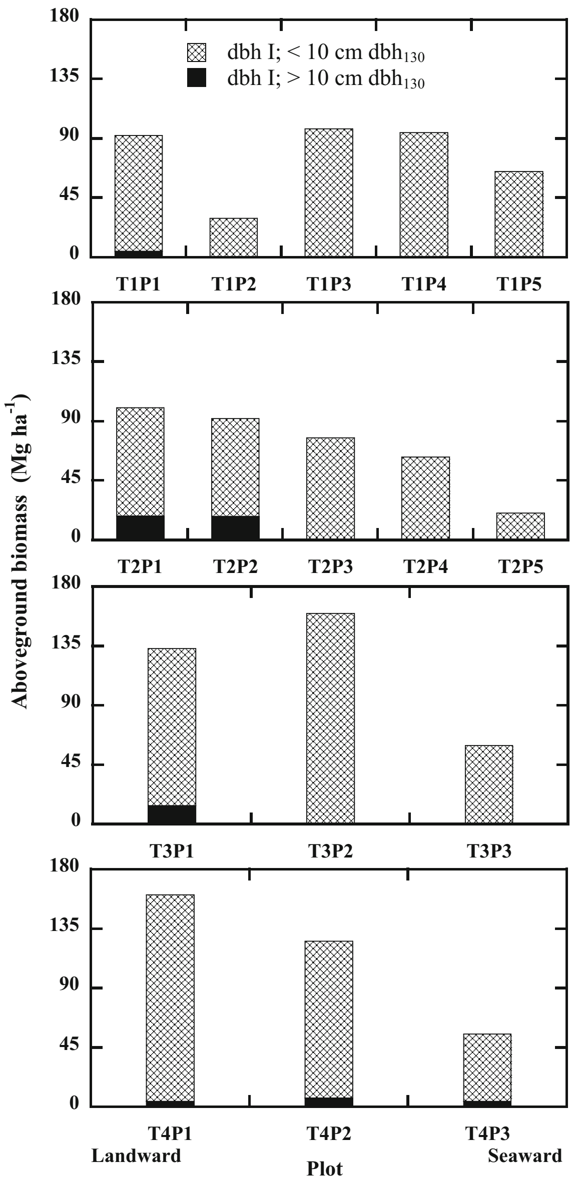


**Fig. 2** Aboveground biomass of all trees (dbh class I = trees < 10 cm dbh; dbh class II = trees > 10 cm dbh) in each plots and transects

et al., 2001; Gross et al., 2014). Finally, the younger seaward plots may show suppressed densities due to natural disturbances such as large waves during storms.

Mean tree diameter range was 4.0–5.8 cm in our study area, which is within the range of 2.5–10.0 cm in 6- and 50-year-old *Rhizophora* plantations in the Philippines (Salmo et al., 2013). Overall mangrove tree diameters generally increase with stand age following self-thinning (Berger & Hildenbrandt, 2003; Deshar et al., 2012; Kamara et al., 2012). However, in our study the 21-year stand had lower tree diameters than a 25-year-old *Rhizophora* spp. stand from Sawi Bay, Thailand and 15-year-old stand from Sarawak, Malaysia (Table 3). This might be due to the stress to seedlings that occurred in these sub-optimal conditions and that resulted in lower tree growth rates and stands that have not reached the point of self-thinning. In our study, the mean annual tree diameter increment ranged from 0.3 to 1.3 cm year$^{-1}$ for the 21- and 3-year stands, respectively. This indicated that, despite these sub-optimal conditions, the younger aged mangrove plantations had faster increases in tree diameter. The average annual diameter increment for *Rhizophora stylosa* Griff. was 0.19 cm year$^{-1}$ from a 10–11-year-old site, 0.13 cm year$^{-1}$ from a 7–8-year-old site, and 0.11 cm from a 3 aged site in Pujada Bay, Davao Oriental, Philippines. These values are all lower than our results, which was likely due to the fact that these mangroves were planted in sites that may have been even less suitable for these species than our sites (Uddin et al., 2014). Other factors that may have influenced annual diameter increment include local abiotic (salinity, rainfall, temperature etc.) and biotic factors (inter- and intra-specific plant competition) (Menezes et al., 2003). Tree heights ranged from 3.8 to 6.2 m for 3- and 21-year-old stands, respectively (Table 3). Earlier reported mean canopy height range from 2.0 to 10.4 m in 6- and 50-year-old aged stands, respectively, in the Philippines (Salmo et al., 2013). The highest height class from our 21-year-old *Rhizophora* spp plantation was 7.16 m (Table 1), which is within the range reported for *Rhizophora* sp. plantations in South East Asia (Srivasatava et al., 1988; FAO, 1993). Present results shows that the majority of trees were smaller in height than dbh class II trees and belonged to the 5–6 cm diameter class, which means this young forest is occupied by small trees (Fig. 2; Supplementary material Fig. 1S). Most of the basal area was accounted for by trees with a dbh of 2.0–9.0 cm. Walters (2000) found that mangrove plantation stands typically have lower basal areas (12.7–21.8 m$^2$ ha$^{-1}$) compared to natural forests (19.4–33.2 m$^2$ ha$^{-1}$). Earlier studies show that basal area does not show any clear relationship with age because net increase in basal area is initially rapid, but as stands get older, the basal area increase rate decreases (Walters, 2000).

**Table 3** Structure and aboveground biomass (ABG) of *Rhizophora* species planted mangrove stands in the different regions

| Location | Species | Age (year) | Tree density ($ha^{-1}$) | Basal area ($m^2$ $ha^{-1}$) | LAI ($m^2$ $m^{-2}$) | Height (m) | Diameter (cm) | ABG (Mg $ha^{-1}$) | References |
|---|---|---|---|---|---|---|---|---|---|
| Sawi Bay, Thailand | *R. apiculata* | 3 | 13,212 | 34.7 | 3.6 | | 4.1 | 65.4 | Alongi & Dixon (2000) |
| Sawi Bay, Thailand | *R. apiculata* | 3 | 9623 | 44.7 | 2.9 | | 4.0 | 46 | Alongi & Dixon (2000) |
| Laguindingan, Philippines | *R. apiculata* <br> *R. mucronata* | 3 | 4903 | 7.3 | 1.7 | 3.8 | 4.0 | 42.6 | This study |
| Sawi Bay, Thailand | *R. apiculata* | 5 | 16,600 | 41.5 | 1.6 | | 3.4 | 47.65 | Alongi and Dixon (2000) |
| Lingayen Gulf, Philippines | *R. mucronata* | 6 | 7780 | | 0.15 | | | 13.58 | Salmo III et al. (2013) |
| Lingayen Gulf, Philippines | *R. mucronata* | 8 | 4244 | | 0.29 | | | 18.21 | Salmo III et al. (2013) |
| Samut Songkarm, Thailand | *R. apiculata* | 9 | | | | 10.24 | 2.61 | 116.81 | Kridiborworn et al. (2012) |
| Laguindingan, Philippines | *R. apiculata* <br> *R. mucronata* | 9 | 4719 | 11.6 | 2.7 | 5.9 | 5.5 | 74.4 | This study |
| Lingayen Gulf, Philippines | *R. mucronata* | 10 | 2387 | | 2.00 | | | 38.45 | Salmo III et al. (2013) |
| Lingayen Gulf, Philippines | *R. mucronata* | 11 | 1886 | | 2.8 | | | 43.29 | Salmo III et al. (2013) |
| Gazi Bay, Kenya | *R. mucronata* | 12 | 4864 | 16.53 | 3.99 | 8.4 | 6.2 | 106.7 | Kairo et al. (2008) |
| Samut Songkarm, Thailand | *R. apiculata* | 12 | 22,089 | | | 11.39 | 3.41 | 140.49 | Kridiborworn et al. (2012) |
| Kalibo, Philippines | *R. mucronata* | 12 | 2122 | | 2.81 | | | 51.43 | Salmo III et al. (2013) |
| Sarawak, Malaysia | *R. apiculata* | 15 | 840 | | | 12.16 | 14.45 | 116.79 | Chandra et al. (2011) |
| Kalibo, Philippines | *R. mucronata* | 17 | 1532 | | 3.91 | | | 101.80 | Salmo III et al. (2013) |
| Lingayen Gulf, Philippines | *R. mucronata* | 18 | 1358 | | 4.13 | | | 115.62 | Salmo III et al. (2013) |
| Matang, Malaysia | *R. apiculata* | 20 | | | | | | 234 | Ong et al. (1995) |
| Laguindingan, Philippines | *R. apiculata* <br> *R. mucronata* | 21 | 5617 | 16.7 | 3.3 | 6.2 | 5.8 | 111.7 | This study |
| Sawi bay, Thailand | *R. apiculata* | 25 | 5402 | 33.3 | 5.1 | | 11.2 | 344.2 | Alongi & Dixon (2000) |
| Bohol, Philippines | *R. mucronata* | 50 | 1231 | | 4.54 | | | 132.18 | Salmo III et al. (2013) |
| Matang, Malysia | *R. apiculata* | >80 | | | | | | 460.00 | Putz & Chan (1986) |

Mangrove LAI ranged between 0.8 and 7.0, with a mean of 3.96 $m^2$ $m^{-2}$ (Green et al., 1997). The 3-year stand in this study had an LAI of 1.7, which was lower than values from the 9- or 21-year-old stands or from 25-year-old *Rhizophora* spp stands in Malaysia that ranged from 2.2 to 7.4 $m^2$ $m^{-2}$ with an average value 4.9 $m^2$ $m^{-2}$ (Clough et al., 1997). This suggests that canopies in the younger stands are still open and start to close with stand development. Forest gaps started to fill in and reach full canopy cover after 11 years in a *Rhizophora* stand in the Philippines (Salmo et al., 2013). The time required for gap filling by Salmo et al. (2013) was much quicker than what was observed in this study. This suggests that other factors, such as the higher salinity and rates of tidal inundation that occurred in the 3-year-old mangrove stands, may have decreased growth and resulted in low-stature scrub forest.

## Changes in biomass (ABG) with age

Combining forest structure data with published allometric equations are often used to calculate biomass as well as carbon stocks (e.g., Donato et al., 2011; Phang et al., 2015; Nam et al., 2016). Trees having dbh < 10 cm contributed the most to ABG measured in the older plots. The significant positive correlation between biomass and stem density as well as basal area suggests that stand structure is closely related to biomass (see also Fromard et al., 1998; Gross et al., 2014). The ABG of *R. mucronata* and *R. apiculata* in different aged stands across the Philippines are shown in Fig. 3, with ABG increasing with forest age. However, the 3- and 9-year mangrove stands planted on seagrass beds from this study had higher ABG compared to other mangrove afforestation projects that occurred on unvegetated sites. Our 21-year stand had lower ABG compared to 18-year planted stands in the Philippines. This suggests that mangrove plantation stands on seagrass areas may initially grow well compared to other unvegetated surfaces (i.e., mudflats), potentially due to the facilitative effects of seagrass protecting mangrove seedlings from hydrodynamics or by changing in situ soil properties (sensu McKee et al., 2007; Friess et al., 2012). However, suboptimal hydrological conditions in the lower intertidal zone over the long-term causes growth to slow down (in the 21-year stand) and eventually collapse in future, as also modeled by Salmo & Juniaco (2015).

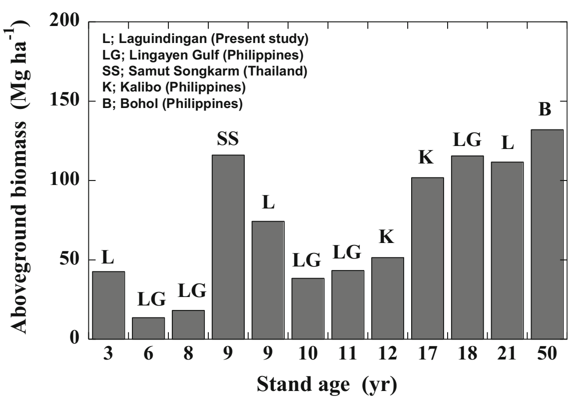


**Fig. 3** Aboveground biomass of different aged stand of *Rhizophora mucronata* and *R. apiculata* across Philippines. *L* Laguindingan (present study site), *LG* Lingayan Gulf, *K* Kalibo, *B* Bohol)

## Impacts of afforestation on seagrass beds

Despite the patterns reported above in mangrove biomass over time, it appears that planting mangrove seedlings in seagrass beds results in the eventual loss of an equally important ecosystem in terms of C sequestration and other ecosystem services (Alongi, 2014). Seagrass cover significantly decreased with the increasing canopy cover of older aged mangrove plots ($r = -0.77, p < 0.001$) (Fig. 4). Seagrass plants were only growing in the presence of mangrove canopy gaps where sufficient sunlight was available for photosynthesis. Ellison et al. (1996) reported that mangrove branches creating light gaps in the canopy allows necessary light to encourage mangrove seedlings, and mangrove seedling growth is increased when aboveground competition and canopies are reduced (Simpson et al., 2013). This is likely why some of seagrass plants were also able to survive in our afforested plots.

Mangrove afforestation on seagrass beds may have implications for ecosystem service provision of both systems. Mangroves are able to sequester more carbon initially due to improved growth on seagrass areas, though this ability is reduced as aboveground biomass decreases in the lower intertidal zone relative to other locations. Reduced seagrass presence also reduces the ability of this ecosystem to sequester and store carbon (Fourqurean et al., 2012; Macreadie et al., 2014). Recent studies suggest that when situated adjacent to mangroves, seagrass meadows had significantly

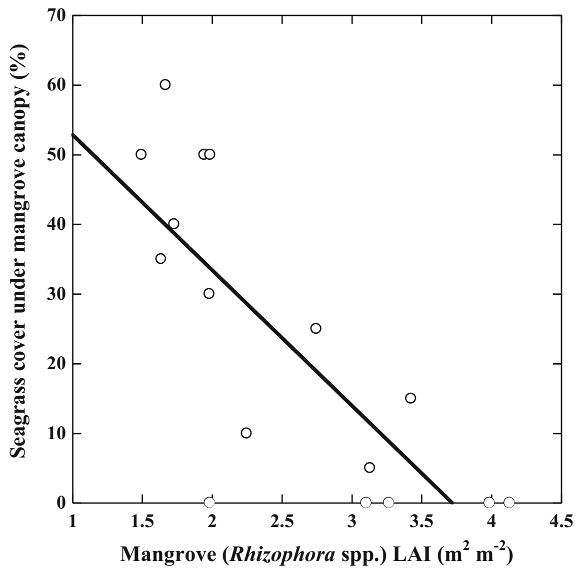


**Fig. 4** Relationship between seagrass cover under the mangrove canopy and mangrove (*Rhizophora* spp.) leaf area index ($R^2 = 0.59$)

higher soil OC concentration than those not situated close to mangroves (Chen et al., 2017). There is significant spatial variation in both soil organic carbon concentration and storage, with values decreasing towards the sea, and the contribution of mangrove-derived carbon to seagrass carbon stocks also reduces with distance from the mangrove forest (Chen et al., 2017). At our site, it is possible that soil organic carbon stocks may increase because of mangrove afforestation on seagrass areas. Mangrove afforestation on seagrass beds also has impacts on other ecosystem services such as fisheries, due to the life cycles of many reef fish species that rely on seagrass beds to complete their life cycles (Nagelkerken et al., 2008). This is already evident from other results from this area that showed differences in fish species composition between mangroves planted on seagrass beds and naturally growing seagrass bed areas (Honda et al., 2013).

Despite the lower intertidal and upper subtidal zone providing sub-optimal hydrological conditions for mangrove establishment and growth, mangroves (particularly *Rhizophora* sp.) continue to be planted on seagrasses and mudflats in the Philippines for a number of reasons. This may be partly due to the fact that government agencies and NGOs are used to using *Rhizophora* spp.. In the past, this species were seen as an economically important forestry resource that was heavily outplanted (e.g., Watson, 1928). *Rhizophora* sp. propagules are also easy to collect and grow into seedling that can easily be distributed to villages for outplanting activities. Planting on seagrass beds and mudflats may also be occurring because of a need for large areas to meet high targets for seedling planting or because this land is easier to access due to a lack of land tenure (Primavera & Esteban, 2008).

Despite these challenges, and even though mangrove afforestation in this study was initially successful in this study due to early facilitation by seagrass, it should be discouraged for two reasons. First, the older planted stands do not appear to be as successful at growth and biomass accumulation compared to other stands at higher elevations. Thus, mangroves should be rehabilitated in areas where the elevation and resulting hydrology is suitable (Lewis, 2005). If land tenure and community use issues can be resolved, the large area of abandoned aquaculture ponds present in the Philippines provides ideal locations for large-scale mangrove regeneration. These areas used to be mangroves and rehabilitation is possible through the careful assessment and restoration of hydrological and geomorphological factors that can support natural regeneration of mangrove seedlings (Matsui et al., 2010; Oh et al., 2017). If there are no mature trees around to provide these seedlings, then outplanting of seedlings should be considered.

Second, lower intertidal/upper subtidal mangrove afforestation can result in the loss of seagrass beds due to shading. Seagrass cover and density significantly decrease where mangrove afforestation has occurred (Rance, 2014). This is of particular concern in the present study site, as it is one of the few extensive seagrass meadows in the Philippines, providing multiple ecosystem services to the local population (Uy, 2001). Therefore, it may not be beneficial to increase one coastal ecosystem at the expense of another.

## Conclusions

Mangrove forest plantations have attracted much attention in last decades from researchers and stakeholders in an attempt to recover ecosystem services lost due to rapid deforestation in Southeast Asia. This study shows that some level of mangrove afforestation is possible, with higher or equal values of stand structural parameters and biomass compared to other planted *Rhizophora* stands of similar age under different elevations relative to mean sea-level rise. Facilitative interactions between coastal ecosystems also suggest that mangrove afforestation on seagrass beds may increase mangrove growth performance, at least over the short term. However, we conclude that mangroves should avoid being planting on seagrass beds, as any benefits are short-term, and afforestation contributes to the loss of ecological functions and services provided by seagrasses. Mangrove afforestation in the Philippines has generally been unsuccessful. Therefore, there is a danger that the planting process and initial mangrove growth may shade out and destroy the seagrass, before itself failing. Future mangrove rehabilitation projects should take a landscape approach that fully considers and accounts for the impact of management activities on adjacent coastal ecosystems.

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**Compliance with ethical standards**

**Conflict of interest** The authors declare that they have no conflict of interest.

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