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Comparative study of mangrove recruitment in oil deforested areas in Guimaras, Philippines

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ABSTRACT

The MT Solar I oil spill that released over 2 million liters of bunker C oil in Guimaras in August 2006 affected mangrove forest in varying degrees from acute damages, sublethal stresses and long term effects. Although the incident caused only <1 ha deforestation, large scale mangrove planting has been implemented as part of response initiatives without inference to scientific assessment of the natural recovery potential of the damaged habitat. The response and intervention in the aftermath of the oil spill primarily involved planting mangroves. This study assessed the mangrove recruitment and colonization in two deforested mangrove areas in Guimaras, Philippines and evaluated whether replanting was necessary as part of intervention. Results on reconstructed age extrapolated from the internodal index showed that recruitment started to take place one and two years after the spill in Site 1 (3.33% of tagged wildings) and Site 2 (11.67%) respectively. However, the wilding establishment in the former site was impaired in the succeeding years and maintained at very low numbers (0.07-0.15 m^{-2}) whereas wilding establishment was high in Site 2 (2.19-3.31 per m^{-2}). The difference between sites is statistically significant ($P<0.0001$). The marked difference between sites is attributed to the prevailing interventions and persistent disturbances. The extraction of dead trees and downed wood in Site 1 impaired the progress of recovery and was further compounded by persistent disturbances brought by the socio-economic activities, whereas, non-intervention in Site 2 favored the progress of natural recovery. Eight years after the incident, the colonizers in Site 2 were already reproductively mature as some already developed flowers. This study highlighted the capacity of mangroves to recover from perturbations as long as the geomorphological features of the habitats are not altered. The large-scale mangrove planting implemented in the affected areas is deemed unnecessary given that one year after the incident recruitment started to progress. Thus, future oil spill responses and interventions should carefully assess the natural recovery processes of the impacted areas first, before proceeding to large scale mangrove planting programs. Otherwise, the good intentions of implementing faster recovery of mangroves and other impacted habitats may be considered a waste of public funds and a futile effort vis-à-vis practicality.

Keywords: oil spill, mangrove mortality, recruitment and recolonization, Guimaras

INTRODUCTION

Mangroves are trees and shrubs that dominate the coastal ecosystem in the tropics and subtropics. The mangrove forest is a highly productive and valuable ecosystem that supports the conservation of biological diversity by serving as habitat, spawning grounds, nurseries and foraging grounds for marine fauna. Mangrove forests also act as natural pollution filters and protect coastal communities. In the rankings of coastal areas in National Oceanic and Atmospheric Administration (NOAA) Environmental Sensitivity Indices, commonly used as a tool for spill contingency planning, mangrove forests are ranked as the most sensitive tropical habitats (NOAA 2014). The sinking of the MT Solar I, carrying more than 2 million liters of bunker fuel occurred during a violent storm approximately 20.4 km off the southwest coast of Guimaras around midnight on August 11, 2006. The spill caused environmental havoc through Guimaras Strait and Iloilo Strait (Figure 1). The incident released more than 2 million liters of Bunker C fuel oil and contaminated over 200 km of coastline – a rich fishing ground, affecting the traditional livelihood of the locals. The incident caused mortality of mangroves that accounted to 0.932 ha (Sadaba et al., 2009a), impacting *Avicennia marina*, *Rhizophora apiculata*, *R. mucronata*, *R. stylosa*, and *Sonneratia alba* (Sadaba et al., 2009b). These species are widely distributed on the island and often found in the seaward margin to the middle zone of the forest; hence, they are more likely vulnerable species to oil contamination.

One initiative from governing bodies when dealing with the aftermath of an oil spill incident is replanting mangroves without conducting baseline site assessments. This was practiced in the case of the Guimaras oil spill, where the concerned government agencies immediately planted *Rhizophora* propagules in areas affected by the oil spill without gathering baseline data on the apparent state and/or allowing for any natural recovery of the affected mangrove habitat. Massive planting efforts have been implemented even in areas not previously occupied by mangroves at the time of the spill. In any restoration effort, the decision to assist in the recovery of the damaged habitat, such as from an oil spill, is based on the assumption that natural processes are inadequate in restoring the damaged habitat (Duke et al., 1999). For this reason, it is necessary to understand the natural recovery of the affected mangroves in order to provide guidance on developing suitable science-based restoration actions. Thus, this study will address the following questions: What is the status of mangrove recovery in the deforested areas three years after the incident? Was there a need for an assisted restoration effort by planting mangroves? When should replanting be implemented as part of an oil spill response and restoration program?

MATERIALS AND METHODS

Study site and establishment of monitoring plot

This study was conducted in two oil spill deforested mangrove areas of Guimaras Island, central Philippines (Figure 1). Monitoring sites were established in Sitio Bagatnan, Barangay Lapaz (Site 1) and in Panobolon Island (Site 2). Both sites are in the Municipality of Nueva Valencia, a worst-hit coastal community in the aftermath of the MT Solar I oil spill. Field sampling was conducted every two months for the first six months and every three months in the succeeding nine months from November 2009 to February 2011 (15 months total). A reference (control) site was established during the first month of the sampling but discontinued in the succeeding months for two reasons: 1.) the total petroleum hydrocarbon in the soil was comparable to the oiled sites, although the mangrove trees in the selected reference site were not physically coated with oil, and 2.) there was no canopy gap in the uncontaminated mangrove that was comparable to the deforested areas caused by the oil spill where studies on recruitment and recovery were conducted. Site 1 is part of the mainland with 0.49 ha deforested, while Site 2 is an islet with 0.39 ha deforested. No planting initiatives have been conducted inside these deforested patches, hence, the pioneer colonizers could be taken as a sign of natural recovery. Common to both study sites was that mortality of mangroves was concentrated in the inward zones of the forest, which are characterized by low hydrodynamics and tidal flushing. However, the two sites have differing anthropogenic activities and socio-economic profile related to human access (Table 1). Site 1 is located near a residential area where people partly rely on fishing and fishing-related activities for their daily subsistence (Figure 2). Site 2 is relatively isolated with no houses in close proximity and the presence of a fishpond contiguous to the mangrove area restricts direct access of locals to the mangrove area.

Recruitment and mortality rate, growth dynamics and period of recruitment

For recruitment (R) and mortality (M) rates, three 5 x 5m (25 m^2) permanent plots were established inside the deforested mangrove patch, and the corners of the plots were demarcated with a PVC pipe. Wildings (<1 m height) and saplings (<4 cm girth and height greater than 1 m) inside the plot were tagged with an improvised flagging marker made of parachute cloth attached to nylon cord. R and M rates were scored every two to three months based on Barnuevo and Sadaba (2014) and Padilla et al. (2004).

For growth dynamics and period of recruitment, 20 wildings per species of *R. apiculata*, *R. mucronata*, and *R. stylosa,* encompassing the full range of heights and girths, were tagged and scored for various physical parameters based on the work of Duke et al. (1999). Metrics included determining heights for all nodes along the main stem from substratum to the topmost or apical hypocotyl node, stem diameter just above the “zero” hypocotyl node, and total number of leafy shoots. For internodal measurements, the sequence of mangrove internodes along the main stem from the zero internode just above the hypocotyl to the growing meristem was measured with a measuring tape. The age structure of the population of *Rhizophora* wildings was obtained by calculating the age of each plant using the reconstructive age determination based on plastochron (age of wildings (in year) = number of internodes / mean annual internode production rate) (Padilla et al. 2004, Duke and Pinzon 1992, Duarte et al. 1999, Ademilua and Botha 2005). Additionally, a visit was made in 2014 to photograph the monitoring plots and to quantify the biomass of downed wood inside the monitoring plots.

RESULTS AND DISCUSSION

Recruitment and density

The density and recruitment values showed a significant difference between sites three years after the incident ($P<0.0001$). Site 2 had an increasing wilding density throughout the study and became stable towards the end of the study period while Site 1 continuously declined (Figure 3). The average density of recruits in Site 1 ranged from 0.07 to 3.3 wildings/m^2 while in Site 2 ranged from 2.19 to 3.31 wildings/m^2. The recruitment and colonization of wildings in Site 1 was impaired. This may be a direct consequence of the removal of dead trees and downed wood. The residents harvested the dead trees in Site 1 and used the logs for firewood, whereas, successful colonization of the forest gap progressed in Site 2 where dead trees and downed wood remained intact (Barnuevo and Sadaba 2014). The removal of downed wood in Site 1 exposed the area to surging waves, and likely subjected the forest gap to a wide range of temperatures, desiccation and erosion from rainfall that consequently changed the substratum (Duke et al 1999). On the other hand, the presence of downed wood in site 2 trapped available propagules and facilitated the colonization of new propagules. In Belize (Central America), mangrove recruitment in a clear-cut forest was accelerated by the presence of *Sesuvium portulacastrum* (a succulent forb) and *Distichlis spicata* (a grass), two coastal species common throughout the Caribbean region (McKee et al., 2007). Had the downed wood remained intact in Site 1, it could have functioned similarly to herbaceous vegetation in assisting the recolonization. Recruitment (R) and mortality (M) rates on the other hand varied spatio-temporally with higher M compared with R in both sites (Figure 4). Among the three species, only *R. apiculata* in Site 1 showed a higher R over M rate. The same observation was also reported by Padilla et al. (2004) in Ulugan Bay, Palawan (Philippines) wherein mortality exceeded rates of recruitment. The observed high M in Site 1 could not be conclusively taken as direct evidence of population decline considering the short study period that ran for only 15 months. The decline of seedling population or high M rate could be compensated in the succeeding years considering a longer period of time.

The production of internodes showed a seasonal trend in both sites with peaks occurring in the month of May and lowest values being reported in the month of November. The average annual production of internodes ranged from 6.8±0.37 to 9.9±0.36 for *R. apiculata* and *R. mucronata* in Site 2 and Site 1, respectively (Figure 5). This observed cyclical pattern implies the existence of both a favorable and an unfavorable time for the production of internodes, attributed to, or mainly driven by, the availability of light that allowed faster food production and therefore node production (Duarte et al., 1999). The same unimodal pattern was also observed in the growth and elongation rate for all the species (Figure 6). A similar pattern was observed by Thi Ha et al. (2003) in Halong Bay (Vietnam) wherein the growth of *Kandelia kandel* showed seasonal differences with high growth rates and faster internode formation in the summer and low growth rates and slower internode formation during the coldest and driest months of the year. The average annual growth rate showed a higher increment in Site 1 compared with Site 2, albeit statistically not significant ($P>0.05$). Among the species, average monthly height increment was highest for *R. mucronata* with 1.8±0.07 cm in Site 1; *R. apiculata* had 1.4±0.06 cm and *R. stylosa* had 1.3±0.04 cm. In contrast, *R. mucronata* in Site 2 had the lowest average monthly height increment of 0.56±0.05 cm, while *R. apiculata* had 0.6±0.04 cm and *R. stylosa* had 0.9±0.04 cm increments. Factors that contributed to the differences in growth rate could be the density of recruits (see Figure 3) and the girth sizes and number of branches per wildings (Figure 7). The high density observed in Site 2 leads to intense competition for space, light, and nutrients which are the determining factors for growth and development of plants, especially for the new recruits. In contrast, the lower density of recruits in Site 1 allowed for ample space, light, and nutrients for the newly established wildings, and thus was favorable for vertical or upward growth. Furthermore, the recruits in Site 2, particularly the species of *R. apiculata,* had significantly higher number of branches compared with Site 1 and relatively larger girths (as shown in Figure 6). In terms of resource partitioning and allocation, the energy production of the wildings in Site 2 was allocated to lateral expansion over vertical growth.

The data on annual node and internode production along the main stem was extrapolated and the age of recruits was reconstructed as shown in Figure 8. All wildings presumably present at the time of the oil spill died with the trees, as was reflected by zero values at the time of the incident. Initial recruitment in Site 1 started one year after the incident with 3.33% (N=2) of the tagged wildings, increased in the second year of the spill with 56.67% (N=34), and then declined in the succeeding year with only 25.0%. In contrast, recruitment in Site 2 began two years after the incident with 11.67% (N=7) of tagged wildings and increased in year three with 61.67% (N=37). It is interesting to note that extraction of dead trees in Site 1 happened at year two, which also coincided with the peak of recruitment. Consequently, the extraction of dead trees led to higher M rates as supported in Figure 4. Presumably, if the dead trees not been harvested, available propagules may have been trapped and may have catalyzed the wildings establishment and recruitment. As a consequence of dead tree removal, there was a significant drop of wilding colonization at year three in Site 1.

Recovery of an impacted ecosystem following a perturbation such as an oil spill is often interpreted to mean a return to the system in place at the time of the spill (NOAA 2010). While considerable debate exists over the definition of recovery and the point at which an ecosystem is said to have recovered, there is broad acceptance that natural variability in ecosystems makes a return to the exact pre-spill conditions unlikely (ITOPF 2011). For instance, mangroves' specialized niche is in a unique, changeable zone, subject to sediment flow that accretes and erodes, varying amounts of fresh water, impacts from storms and hurricanes, invasion by foreign species, and predation. Thus, even if there is a precise description of ecosystem conditions just before the spill, reversion to the pre-spill state remains uncertain (NOAA 2014). Most definitions of recovery instead focus on the re-establishment of a community of flora and fauna that is characteristic of the habitat and functions normally in terms of biodiversity and productivity. According to Duke et al. (1998), recovery of deforested sites depends chiefly on site exposure. In exposed locations, sediment flushing appeared important in stimulating recovery, while greater exposure also allowed premature destruction of growing plants by strong seas and storms. Some oiled areas may recover after 30 years depending on the time it takes for trees to reach maturity and on the substrate condition (Burns et al., 1993, Jacobi and Schaeffer-Novelli, 1990).

When should replanting be an option?

The most common initiative from governing institutions in dealing with the aftermath of an oil spill incident, particularly in the Philippines, is to plant mangroves immediately without conducting baseline surveys. In any oil spill restoration effort, the decision to assist in the recovery of the impacted habitat is based on the assumption that natural processes are inadequate in restoring the damaged habitat. Thus, before proceeding to mangrove restoration, it is vital to have knowledge of the natural processes and recovery dynamics of the impacted areas. Assessment of recruitment, mortality and growth of seedlings in the deforested and or impacted areas can serve as basic indicators. Otherwise the good intentions of aiding in the recovery process could result in further destruction of the already disrupted community.

Since each oil spill incident is unique, it is necessary for responders to have baseline information and technical knowledge of the affected habitat when implementing response protocols and interventions. For mangroves affected by an oil spill, understanding the vegetation dynamics is important for conservation, restoration and sustainable exploitation purposes (Santos et al. 2012). The information on mangrove dynamics serves as a basis for deciding whether or not human interference in the form of management or restoration is appropriate (Dahdouh-Guebas et al. 2004). Response to oil spills should seek to minimize the severity of the environmental damage and to hasten the recovery of any damaged ecosystem; and should always seek to complement and make use of natural forces to the fullest extent possible (Orimoogunje and Ajibola-James 2013).

The data on recruitment, mortality, and growth showed a remarkable difference between the two sites. We attributed these results to differing interventions and anthropogenic activities. The area in Site 1 is in close proximity to a residential area where downed woods were harvested, and further compounded by recurring disturbances from various socio-economic activities. The removal of downed wood created forest gaps, subjected the area to a wide range of temperatures, desiccation, and erosion that consequently changed the nature of substratum and impaired the recovery.

The large-scale mangrove planting in Guimaras covering several hundreds of hectares (Burgos 2006) is deemed unnecessary since natural recovery was already in progress one year after the incident in Site 1, and two years after in Site 2. An old adage says that "nature has the capacity to heal itself". In particular, mangroves are considered to be resilient ecosystems that can recover if the geomorphological and hydrological features of their habitat are not changed by human use (Martinuzzi et al. 2009). Thus, during any oil spill response and intervention, careful assessments should first be made regarding natural recovery processes before proceeding to mangrove planting. Otherwise, the good intention of aiding in expedited recovery may result in a waste of public funds and efforts.

CONCLUSION AND RECOMMENDATIONS

This study highlighted the recruitment and natural recovery processes in two deforested mangrove areas with varying interventions and anthropogenic activities in the aftermath of the MT Solar I oil spill. Recruitment and natural recovery of the deforested patches started to take place one and two years after the oil spill in Site 1 and Site 2, respectively. However, the recruitment and wilding colonization in the former site was arrested and maintained at relatively lower values compared with the latter site, where recruitment continued to progress towards the end of the study period. This could be attributed to the persistence of disturbances related to varying anthropogenic activities. Site 1 was located near a residential area where the local population subsists on fishing and fishing related activities, whereas Site 2 was isolated from the community and the presence of a fishpond restricted access of the locals to the mangrove area. The extraction of dead trees in Site 1 two years after the oil spill adversely impacted the progress of recruitment and colonization, and consequently resulted in a decline in the density of young recruits. The removal of dead trees (in Site 1) created forest gaps, subjected the area to a wide range of temperatures, desiccation, and erosion that consequently changed the nature of substratum and impaired the recovery. As a result, the available propagules could not successfully colonize the deforested area, hence, recruitment and recovery in Site 1 was impaired. In contrast, the presence of dead trees in Site 2 (no extraction), trapped some propagules, which kept them from drifting away, and the presence of dead trees and downed wood served as structural support among the recruits, therefore facilitating the colonization of new cohorts. In this study, non-intervention and protection of the damaged area (in the case of Site 2) favored the progress of natural recovery in the oil spill deforested area.

It has been widely accepted that nature has ways and the capacity to heal itself from perturbations, particularly if the geomorphological features are not altered. Future response initiatives pertaining to oil spills should consider natural recovery processes before proceeding to assisted recovery. Technical assessment and monitoring of the natural recovery processes of the impacted areas are recommended. Outlined below are the recommended guidelines based on this study and others (Duke 2016, NOAA 2014, Duke et al. 1999, and Cintron and Schaeffer-Novelli 1983), which could be helpful for policy makers and response managers responsible for mitigating the impacts of oil spills and habitat management and rehabilitation.

1. Quantification of oiling and acute impact assessment. The extent of oiling should be mapped out immediately to identify priority areas for response initiatives. Short term (up to three months) or acute impacts that involve chlorosis or yellowing of leaves, defoliation and mortality should be carefully assessed. If mortality is observed, aerial photographs should be taken to survey the deforestation. In the case of M/T Solar I, hydrocarbon in the sediments and biological samples were quantified immediately after the incident and monitored at 3, 6 and 12 months post-spill (Pahila et al. 2010a and b, Barnuevo and Sadaba 2014).

2. Minimize additional disturbances. Mechanical removal of oil and clean-up operations should be carried out properly to minimize additional impacts. Mangroves are an intricate habitat and are hard to clean. Oil attached to mangroves and stranded in the mangrove substrate may be better left untouched. Where clean-up inside mangroves is deemed necessary, wildings and roots should not be cleared and the dead trees or downed wood should not be removed. Downed wood traps available propagules, serves as structural support, and facilitates recruitment. Additionally, heavily impacted areas, for instance the deforested areas, should be protected from further disturbances like boat parking and other socio-economic activities.

3. Establish monitoring sites and evaluate natural recovery. Natural recovery potential needs to be evaluated before attempting to aid in the recovery. This entails establishing permanent scientific monitoring plots and tagging wildings in the impacted site. In addition, a reference site (not affected by oil) that is comparable to the impacted areas should also be established. Recruitment, mortality and growth of wildings could be used as an indicator. Additionally, regular quantification of sediment hydrocarbons, monthly in the first six months and quarterly in the succeeding months up to two years, is deemed important to determine its degradation over time.

4. Where necessary, plant appropriate mangrove species in suitable zone. Planting should only be considered if the natural recovery processes are retarded. A minimum of one to two years of observation should be undertaken prior to planting, and the levels of the PAH in the sediments should be monitored. Mangroves should only be planted in areas in which they previously occupied.

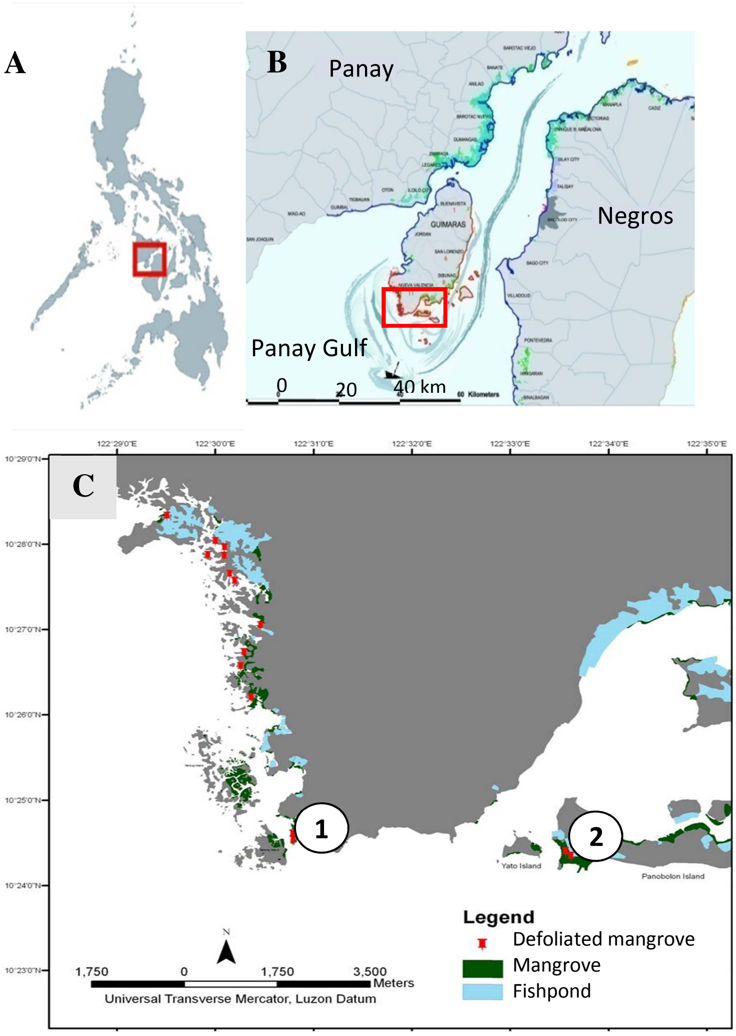

Figure 1. Map of a.) the Philippines, b.) Guimaras Island, and c.) study sites (in circle). 1 - Site 1 (Bagatnan); 2 – Site 2 (Panobolon Island)

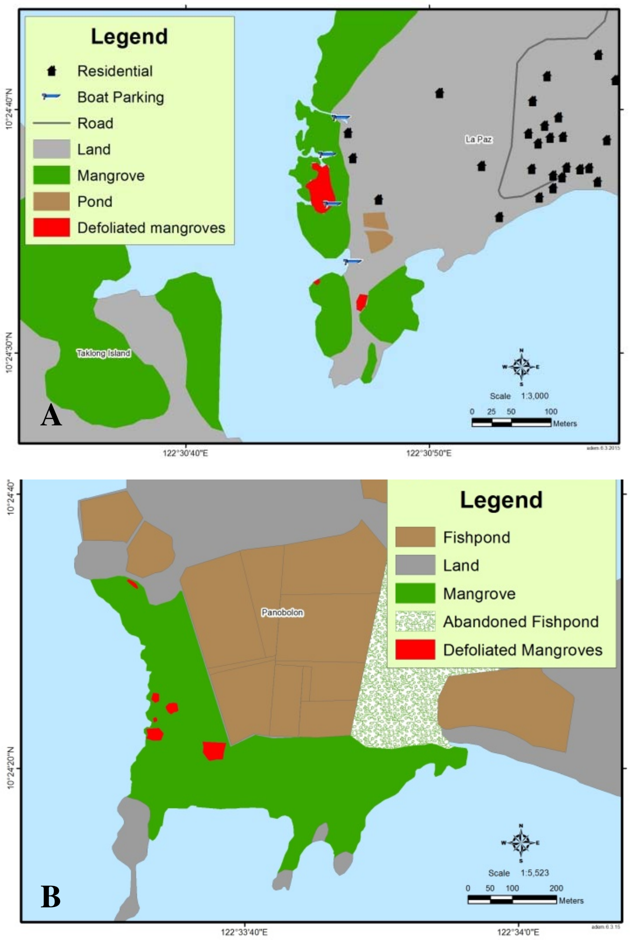

Figure 2. Map showing the socio-economic activities in a.) Site 1 and b.) Site 2. Map by Allan Moscoso

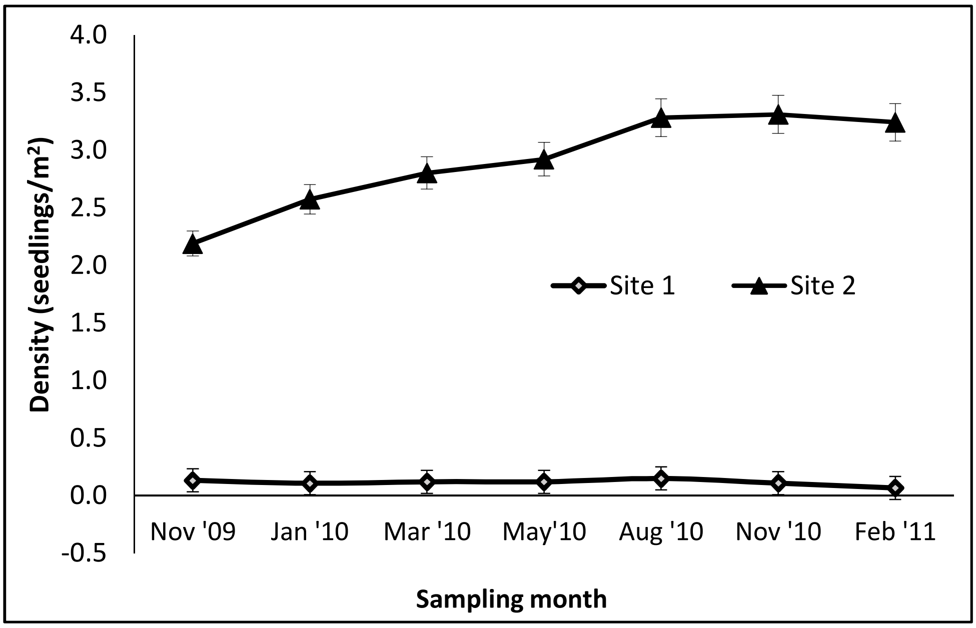

Figure 3. Average wildings density (plants/m^2) inside the monitoring plots from both sites. ($P < 0.0001$)

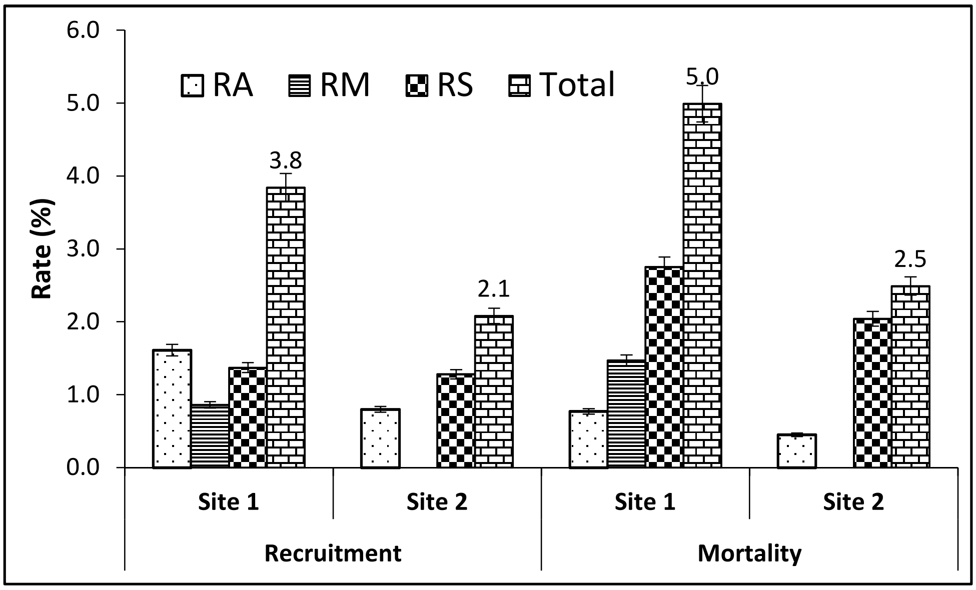

Figure 4. Average annual recruitment and mortality rate of three tagged species of *Rhizophora* in the deforested areas. (RA – *R. apiculata*, RM – *R. mucronata*, RS – *R. stylosa*). (±s.d., $P > 0.05$)

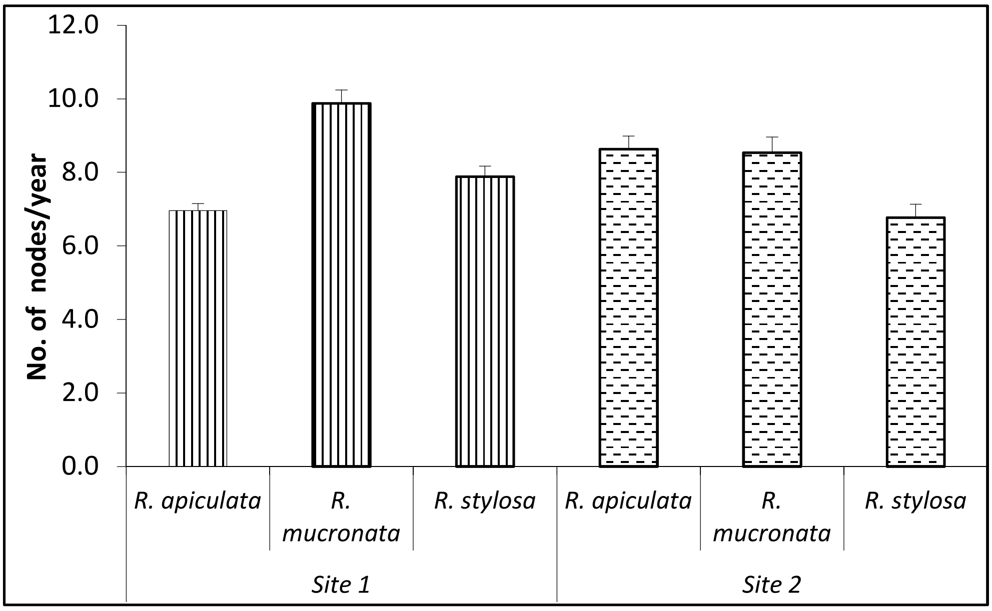

Figure 5. Average annual internode production of tagged wildings (n=20 per species per site) in both study sites. (±s.d., P > 0.05)

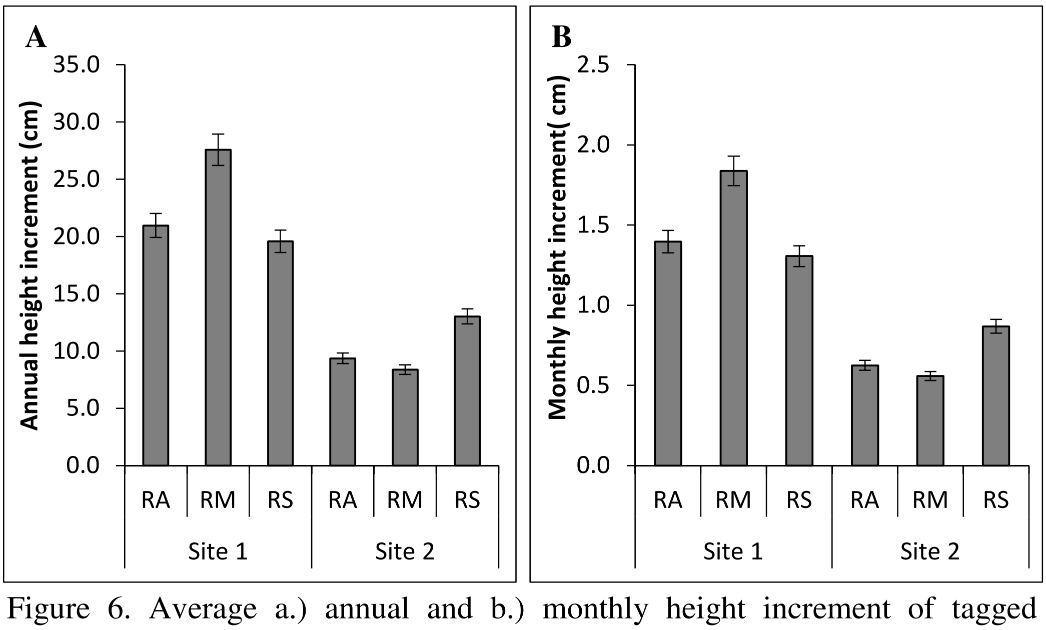

Figure 6. Average a.) annual and b.) monthly height increment of tagged wildings (n=20 per species per site) in both sites (±s.d., P > 0.05). (RA – *R. apiculata*, RM – *R. mucronata*, RS – *R. stylosa*).

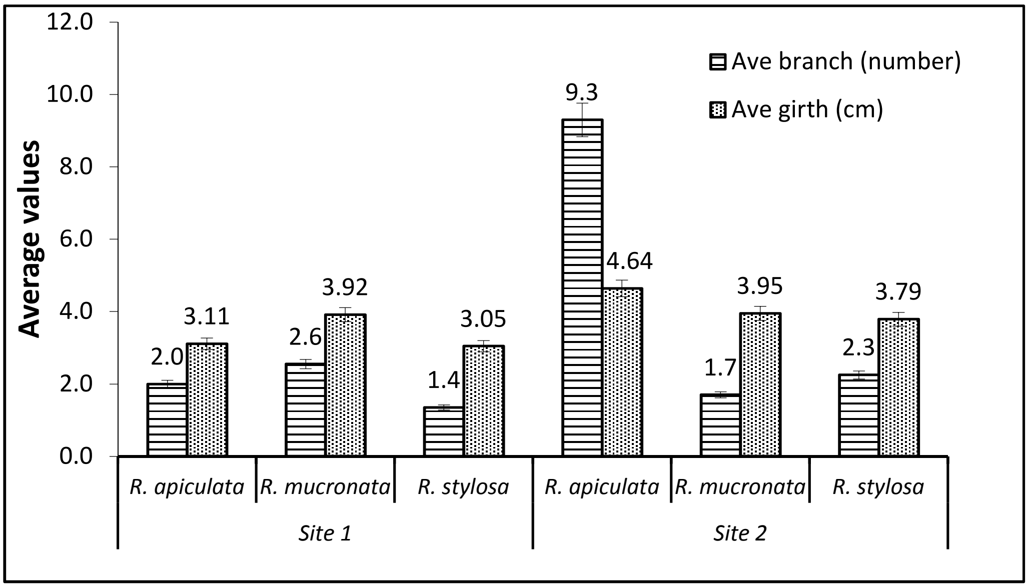

Figure 7. Average number of branches and girth size (cm) of tagged wildings (n=20 per species per site). (±s.d., $P > 0.05$)

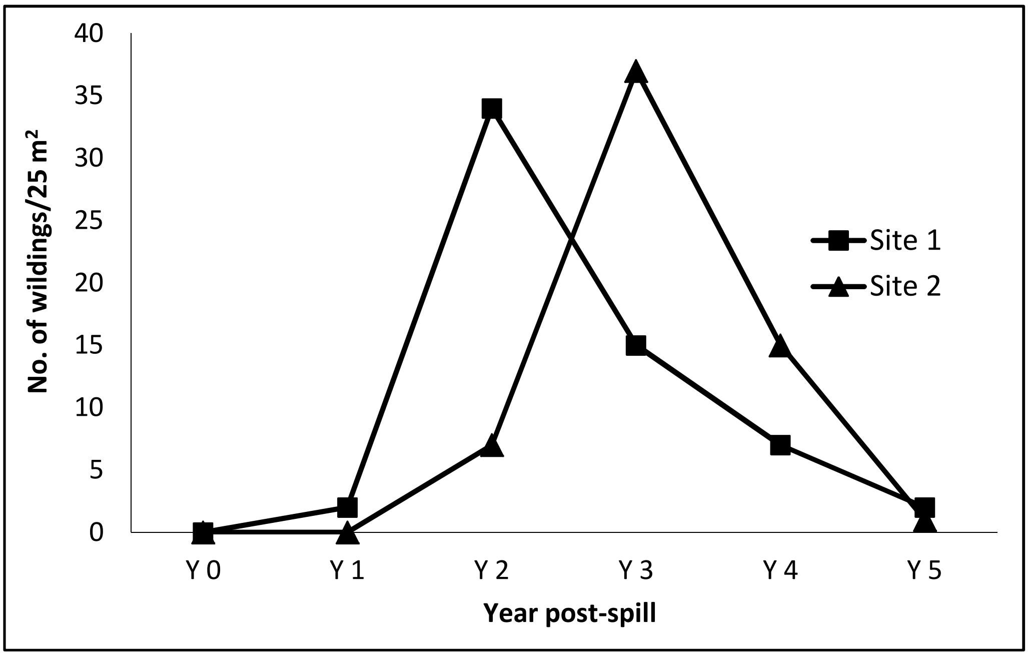

Figure 8. Reconstructed recruitment period of wildings inside the deforested area. ($P < 0.05$)

Table 1. Socio-economic profile and management interventions.

Observations	**Site 1 (Bagatnan)**	**Site 2 (Panobolon)**
Number of houses (within 0.5 km proximity)	11	None
[a]Number of boats parked (average)	9	0
Fishpond area (ha)	<0.09	>2
Economic activities	[b]gleaning, boat parking	pond operation restricted
	near marine reserve	privately owned
Downed wood harvested (intervention)	Yes (year 2)	No
Wet weight of downed wood	0	6.0±2.3 (kg/m^2)
Replanting (as response)	[c]Yes (*Rhizophora* sp.)	No

[a] number of boats was based on the interview with the fisherfolks

[b] gleaning - a fishing method used in shallow coastal area exposed during low tide

[c] planting was conduted outside the study site (~500 m away)

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