Diversity of native woody regeneration in exotic tree plantations and natural forest in Southern Philippines

Adrian M. Tulod, Jupiter V. Casas, Rico A. Marin and Jocyl Ann B. Ejoc

College of Forestry and Environmental Science, Central Mindanao University, University Town, Musuan, Maramag, Bukidnon 8710, Philippines

ABSTRACT

The use of exotic species in reforestations is one of the highly criticized forest policies in the Philippines, mainly due to their perceived negative impacts on biodiversity conservation. To ascertain the influence of exotic plantations on native flora establishments, we inventoried the structure, composition, and diversity of understory woody regeneration in three exotic stands in Southern Philippines and compared them to adjacent second growth forest. The mean total density of regeneration did not differ significantly among the stands, except for the separate density of saplings and seedlings where natural forest had significantly the lowest and highest density, respectively, over the exotic stands. Teak (Tectona grandis L.) and mangium (Acacia mangium Willd.) stands generally had the bigger basal area, indicating the dominance of saplings in these areas. A low diversity characterized the four stands with the lowest and highest diversity indices observed, respectively, in the mahogany (Swietenia macrophylla King) and natural forest. Despite their proximity, each stand exhibited uniqueness in species composition, with some of the endangered species observed only in the exotic stands. Therefore, it would be interesting to know how continued protection of these stands would affect the trajectory of succession of native species over time.

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Introduction

While natural forest losses occur continuously (e.g. from 4128 million ha in 1990 to 3999 million ha in 2015), the extent of forest plantations has increased worldwide by over 110 million ha since 1990, accounting for 7% of total forest area (MacDicken et al. 2015). The second largest area of these plantations is found in the tropics with 57 million ha, an increase of 69% over the last 25 years (MacDicken et al. 2015). Forest plantations established through planting and/or deliberate seeding in the process of forestation are often characterized as even-aged monoculture stands with a broad range of management intensity, but often with short rotation periods especially for plantations with commercial use (FAO 2010). In 2010, 117 countries, representing 67% of the total global forest area, reported using introduced or exotic species in planted forests (FAO 2010).

In the Philippines, where intense population pressure along with limited economic opportunities are major drivers of forest degradation, the use of fast growing species with high economic value has been a popular choice in previous national forest rehabilitation programs. Although they constitute a significantly small area or c. 5%–6% of the total forest cover, mainly due to failed reforestation projects in the past, tree plantations of fast growing exotic species such as mahogany (Swietenia macrophylla King), falcata (Parasenianthes falcatoria (L.) Nielsen), yemane (Gmelina arborea Roxb.), Acacia spp., and teak (Tectona grandis L.) have become an important resource for the country’s pulp and timber needs, thus providing an immediate alternative to ease the pressure on remaining natural forests. Large areas of these plantations are located in Southern Mindanao Island, which is considered the forerunner in industrial tree plantation development in the country (Paler et al. 1998).

While the use of fast-growing tree species in reforestation has generally yielded positive economic, environmental, and employment benefits (Niskanen and Saastamoinen 1996), their impacts on biodiversity conservation and recovery have been viewed negatively especially where exotic species are involved. Exotic species can be difficult to control and may interfere with restoration projects. They have the potential to alter ecosystem functions and soil attributes, which can make restoration sites inhospitable for the re-creation of native ecosystems and often contribute to the demise of native species (D’Antonio and Meyerson 2002; Norton and Forbes 2013). However, as the future demand for industrial wood products is projected to rise by at least 25% from 1996 levels (see: Whiteman and Brown 1999, for a comprehensive forecast of the supply and demand for wood and wood products), an improved understanding of their potential conservation value or influence on native biodiversity would be critical to the development of ecologically and socially sustainable policy measures (Goldman et al. 2008). This can actually help reduce management complications surrounding the purposeful use of exotic tree species in restoration projects (D’Antonio and Meyerson 2002).

The use of native species in reforestation is obviously a sound restoration approach as native forests are the more favorable habitat for biodiversity conservation than exotic forest plantations. However, there is ample evidence that tree plantations can provide favorable conditions for biodiversity conservation either as habitat or nurse plants for native and threatened species establishment (e.g. Lugo 1992;
Chapman and Chapman 1996; Parrotta, Turnbull, et al. 1997; Brockerhoff et al. 2008; Pawson et al. 2010; Becerra and Montenegro 2013) comparable to that of natural forests (e.g. Lugo 1992; Fimbel and Fimbel 1996; Carnus et al. 2006; Koonkhuenthod et al. 2007). Even the widely invasive exotic tree *Pinus radiata*, for instance, has been found to facilitate regeneration of native species in a semi-arid ecosystem in central Chile, and has been recommended in areas where nurse vegetation to facilitate native regeneration is not available (Becerra and Montenegro 2013). In the Philippines, however, published quantitative data to demonstrate the facilitative role of exotic tree plantations on indigenous species establishment is still very limited, although the information is critical in deciding whether tree plantations can be a valuable component in the country’s indigenous forest rehabilitation and conservation programs.

The main aim of this study was therefore to ascertain whether exotic tree plantations can be valuable habitats for indigenous flora establishment. This study is, however, small scale as we only examined the effects of a few existing exotic tree plantations in one geographic area on native woody species regeneration. Specifically, we inventoried the diversity and composition of native woody regeneration in the three existing tree plantations in Southern Mindanao and compared them to adjacent second growth natural forest in the area. Permanent sample plots were used in each stand to facilitate long-term monitoring of native woody regeneration and recruitment, and hence the present study will serve as a baseline for future measurements.

**Methodology**

**Study site**

The study was conducted at the forest reserve managed by Central Mindanao University (CMU) in Musuan, Maramag Town, Province of Bukidnon, Philippines (7.8649°N, 125.0509°E) consisting of second growth natural forest and planted stands of mahogany (*Swietenia macrophylla* King), teak (*Tectona grandis* L.f.), and mangium (*Acacia mangium* Willd.) (Figure 1).

As previously described in the study of Tulod (2015), these tree plantations were established by CMU to reforest areas dominated by *Imperata cylindrica* (L.) Raeusch within its landholdings, which later became subject to selective cutting for the university’s mini-sawmill. Of these stands, mahogany is the oldest as this was established in 1985; while mangium and teak stands were established in 1999 and 2000, respectively. Each stand covers an area of at least 10 ha and had an initial spacing of 2 m x 2 m. The second growth forest is estimated to be more than 30 years old, being one of the most heavily logged areas in Mindanao until the year 1980. Teak and mangium stands are located close to the second growth forest at about 100 m and 300 m away from its

![Figure 1. Location of the four plantations in Musuan Town, Maramag, Bukidnon, Philippines.](image-url)
edge, respectively; while mahogany is located about 2 km away from the natural stand boundary. At least 1 ha in each of these second growth forest and reforestation stands was delineated in 2012 and are being used as sites for ecological related researches. Sampling was carried out 1 year after plot delineation. Although a yemane (Gmelina arborea Roxb.) plantation is also located in the area, this was not sampled as the delineation was not yet completed during the conduct of the study. Since all stands in the study were formerly Imperata grasslands, differences in composition and diversity of understory regeneration should have been due to the successful dispersal of different species from the natural forest in the vicinity and the ability of each stand to support succession.

The Province of Bukidnon is geographically located between 07° 25’–8° 38’N and 124°03’–125°16’E at an average elevation and slope gradient of 915 m and 2938 m above sea level, respectively (Bukidnon 2016). Based on the modified Coronas classification, the climate falls under type IV with no dry season and rainfall more or less evenly distributed throughout the year (Bukidnon 2016). Mean annual rainfall in the province is 2800 mm, while mean annual temperature and relative humidity ranges are 20°C–34°C and 90.86%–92.85%, respectively (Bukidnon 2016).

**Sampling of regenerations**

Ten sample plots measuring 10 m × 10 m were located within the 1 ha permanent monitoring plot of each stand and were established in the direction that represented the characteristics of each site (i.e., in terms of stand structure and topography). Total sampling area was 1000 m² for each site. In each plot, all indigenous saplings (i.e., trees with height > 1.3 m but ≤ 15 cm diameter at breast height [dbh]) found were counted, identified to species level, and measured for their dbh. Enumeration of all woody seedlings with height from 0.3–1.3 m were carried out inside a 3 m × 3 m sampling frame established in the center of each plot. Seedlings with height below 0.3 m were not included in the study as they were difficult to identify and are known to have very high mortality (Otsamo 2000). The diameter was measured at 5 mm above the ground for seedlings and at dbh or 1.3 m above the ground for saplings.

A taxonomy expert at the College of Forestry and Environmental Science (CFES), CMU was consulted on species identification and all the species were further verified using the CFES herbarium. The mode of dispersal of each species was determined based on the actual observations, local knowledge, and literature review. The International Union for Conservation of Nature (IUCN) Red List of Threatened Species (IUCN 2016) and the threatened plants of the Philippines by Fernando et al. (2008) were used to verify the conservation status of the species in the study.

**Data analysis**

The species richness and total species richness were computed as the number of species in each sample plot and the total number of species at each study site, respectively. To see the diversity of species and how the study sites grouped together based on similarity of regeneration, data were subjected to Shannon-Wiener index (H) and Bray-Curtis cluster analysis, respectively, using BioPro software (McAleece et al. 1997). The Shannon-Wiener indices (H) obtained were then converted to effective number of species (ENS) (i.e., the number of equally-common species required to give a particular value of an index according to Jost 2006). ENS allows robust comparison and interpretation of species diversity indices, and was calculated as an exponential of the Shannon index (Jost 2006):

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ENS = \exp(H)
\]

The modified species importance value (SIV) was used to determine the dominance of a species in the study sites (Parrotta, Turnbull, et al. 1997). This was determined by summing up the percentage values of the relative density and relative frequency. The relative density is the density of a species divided by the density of all species and multiplied by 100. The density of a species was calculated as the number of individuals of that species per hectare. The relative frequency is the frequency of a species divided by the total frequency of all species and multiplied by 100. The frequency of a species was determined in terms of the number of sample plots in which that species was found.

All statistical analyses were undertaken using R software (R Development Core Team 2015). Since count data (density and species richness) were highly overdispersed and non-normal, we used a generalized linear model (GLM) specifying negative binomial distribution with log link (using the R MASS package) for all regeneration density and seedling density data as it indicated better goodness-of-fit with a chi-square test (based on the residual deviance and degrees of freedom) when compared with the GLM Poisson regression. Zero-inflated negative binomial (ZINB) regression with log link (using the R pcr package) was fitted for sapling density and species richness as GLM models (Poisson and negative binomial) did not fit the data well via chi-square test. ZINB is recommended for count data with significantly more zeros than expected for a Poisson or negative binomial distribution, and with significantly overdispersed (based on estimated theta parameters) count outcome variables in the zero-inflated model (Zuur et al. 2009). For basal area (BA), the Shannon-Wiener diversity index, and ENS we used one-way ANOVA (with log transformation to account for the large variation in the standard deviations) or Welch’s ANOVA when log transformation did not solve the heteroscedasticity issue. The Welch’s ANOVA is especially recommended for balanced design with very large variation in the standard deviations (McDonald 2014). Significant results (\(\alpha \leq 0.05\)) were compared using the glht function of the R multcomp package for GLM and ANOVA models and least square means of the R lsmeans package for zero-inflated model. Games-Howell post hoc test was used for significant results (\(\alpha \leq 0.05\)) in Welch’s ANOVA model.

**Results**

**Composition of native woody species regeneration in the four stands**

A total of 46 indigenous tree species and 421 individuals from 25 families were recorded in the four study sites (Table 1). Based on the ranking of species importance value (SIV), the dominant species varied among stands. Natural forest was dominated by Knema glomerata, Clausena brevistyla, and Lansium domesticum; mangium plantation by Micromelum compressum var. inodorum, Melanolepis multiglandulosa, and Shorea contorta; teak plantation by Semecarpus

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Families Lauraceae, Meliaceae, and Rutaceae were abundant respectively, of 31, 67% with each of these families composed, Sapindaceae (16%), Rutaceae (12%), Anacardiaceae (12%), and Meliaceae (12%) with each of these families composed, respectively, of four, five, and three mostly pioneer species. Families Lauraceae, Meliaceae, and Rutaceae were abundant in the natural forest, while Anacardiaceae and Sapindaceae were dominant in the teak and mangium tree plantations, respectively. No dominant family was observed on the mahogany stand as the three families observed were represented by only one species.

In terms of the number of stems or individuals, the highest was observed under the natural forest with 236 stems, while the lowest was from mahogany stand with 42 stems. The predominant form of seed dispersal of native species regeneration in the four stands was by animals. Only three species were wind-dispersed: Wrightia pubescens and the two critically endangered species, S. contorta and P. indicus forma echinatus. Most of the species (31, 67%) have unclassified conservation status (UN) based on the IUCN red list category; while 15 species (33%) are threatened under the following categories: least concerned or lower risk category (seven), vulnerable (four), endangered (two), and critically endangered (two).

The seven most dominant species across the four stands, ranked by density and comprising 53% of the total number of individuals, were 

| Botanical name | Family | Number of stems | Species importance value |
|----------------|--------|-----------------|--------------------------|
| Aegia tomentosa Teijm & Binn. | MELIACEAE | 42 | 94 |
| Antedema ghaseembila Gaertn. | PHYLANTHACEAE | 46 | 236 |
| Artocarpus odontosimus Blanco | MORACEAE | 200 | 200 |
| Artocarpus odoratissimus Blanco | MORACEAE | 200 | 200 |
| Bellschmiedia cairocan Vidal | LAMIACEAE | 200 | 200 |
| Bridelia penangiana Hook.f. | LAMIIACEAE | 200 | 200 |
| Buchania arborescens (Blume) Blume | ANACARDIACEAE | 200 | 200 |
| Calophyllum blancoi Planch. & Triana | CLUSIACEAE | 200 | 200 |
| Cananga odorata (Lam.) Hook.f. & Thomson | ANNONACEAE | 200 | 200 |
| Canarium asperum Benth. | BURSERAEEAE | 200 | 200 |
| Celtis philippinensis Blanco | ULMACEAE | 200 | 200 |
| Chrysophyllum cainito | SAPINDACEAE | 200 | 200 |
| Cryptocarya amara | VITACEAE | 200 | 200 |
| Cryptocarya ampla Merr. | LAURACEAE | 200 | 200 |
| Cryptocarya sphenopus Merr. | SAPINDACEAE | 200 | 200 |
| Cryptocarya sphenopus Merr. | ULMACEAE | 200 | 200 |
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by only one stem. Hierarchical cluster analysis of the four stands showed that, despite their proximities or having similar geographical characteristics, the four stands exhibited uniqueness in terms of species composition with percent similarity index of only c. 15.91% (two species) between mahogany and mangium, 15.38% (four species) between mangium and teak, and 10.81% (five species) between teak plantation and natural forest (Figure 3). No similarity in species composition between mahogany and natural forest was observed, while only about 5% (two species) of species in mangium was similar to natural forest.

However, separating regeneration in terms of numbers of sapling (zero-inflated negative binomial regression $\chi^2(3) = 7534.7, P < 0.001$) and seedling (generalized linear regression $\chi^2(3, 36) = 46.69, P < 0.001$) categories in each stand yielded significant differences among the four stands (Figure 5).

### Density and basal area of native woody species regeneration

The mean density of all native tree species regeneration (i.e. the combined number of seedlings and saplings) did not differ significantly (generalized linear regression $\chi^2(3, 36) = 48.98, P = 0.1897$) among the four stands (Figure 4). But numerical values indicated that mangium plantation favors a greater number of individuals with $1570 \pm 729$ stems ha$^{-1}$ followed closely by the natural forest ($1283$ stems ha$^{-1} \pm 619$), then mahogany ($595$ stems ha$^{-1} \pm 295$), and teak stand ($276$ stems ha$^{-1} \pm 65$).
The three exotic tree plantations were significantly \((P < 0.001)\) higher in sapling density than the natural forest, although significant differences \((P < 0.001)\) were also observed among the three exotic stands. Among the stands, mangium significantly tallied the highest number of saplings with \(1556 \pm 732\) stems ha\(^{-1}\), followed by mahogany \((519 \pm 299\) stems ha\(^{-1}\)), teak \((26 \pm 7\) stems ha\(^{-1}\)), with the least number from natural forest with only \(5 \pm 3\) stems ha\(^{-1}\). In contrast, the natural forest had remarkably the highest density of seedlings with \(1278 \pm 619\) stems ha\(^{-1}\). This was significantly higher \((P < 0.001)\) than the density of seedlings observed under the mangium \((14 \pm 7\) stems ha\(^{-1}\)) and mahogany \((76 \pm 37\) stems ha\(^{-1}\)) stands; although not statistically different from the teak stand \((P = 0.1055)\), which had an average of \(250 \pm 62\) stems ha\(^{-1}\). A significant difference in seedling density was also observed between teak and mangium stands \((P < 0.001)\); while teak versus mahogany \((P = 0.3468)\) and mahogany versus mangium \((P = 0.0805)\) had no significant difference.

On the other hand, there was significant variation in the mean (BA) of all native regeneration in the four stands (one-way ANOVA \(F_{[3,36]} = 5.63, P < 0.01\) mainly due to significant differences between teak and mahogany \((P = 0.0053)\) and between natural forest and mahogany stands \((P = 0.0295)\) (Figure 6). No significant differences \((P > 0.05)\) in BA were observed in the following: mangium versus mahogany; mangium versus natural forest; and teak versus mangium and natural forest. Numerically, however, teak had the largest stem sizes at \(20.61 \pm 5.16\) m\(^2\) ha\(^{-1}\), which is almost twice the sizes of those under the natural forest \((11.61 \pm 2.07\) m\(^2\) ha\(^{-1}\)) and mangium stand \((11.73 \pm 5.62\) m\(^2\) ha\(^{-1}\)), and more than five times those under the mahogany stand \((3.38 \pm 1.57\) m\(^2\) ha\(^{-1}\)).

Similarly, significant results were observed for the separate mean BA of saplings (one-way ANOVA \(F_{[3,36]} = 3.0847, P = 0.03936\)) and seedlings (one-way ANOVA \(F_{[3,36]} = 37.12, P < 0.001\)) in the four stands (Figure 7). The natural forest had a significantly greater BA of seedlings at \(8.19 \pm 1.16\) m\(^2\) ha\(^{-1}\) as compared to teak \((3.70 \pm 0.85\) m\(^2\) ha\(^{-1}\), \(P < 0.01\)), mahogany \((0.86 \pm 0.39\) m\(^2\) ha\(^{-1}\), \(P < 0.001)\), and mangium \((0.22 \pm 0.12\) m\(^2\) ha\(^{-1}\), \(P < 0.001)\) plantations. The seedling BA of the latter two exotic plantations, however, was not statistically different \((P = 0.48)\); while highly significant differences \((P < 0.001)\) were noted between the plantations of teak and mahogany, and between teak and mangium. For the saplings, significant difference in mean BA was noted only between teak and mahogany plantations \((P = 0.0456)\). Numerically, the sapling BA followed a trend: teak \((16.91\) m\(^2\) ha\(^{-1}\)) > mangium \((11.51\) m\(^2\) ha\(^{-1}\)) > natural forest \((3.42\) m\(^2\) ha\(^{-1}\)) > mahogany \((2.52\) m\(^2\) ha\(^{-1}\)).

**Richness and diversity of native woody species regeneration**

The differences in species diversity, richness, and the equivalent ENS among the four plantations are shown in Table 2. The native woody regeneration, both under exotic tree plantations and natural forest, was characterized by a very low
species richness and diversity. As expected, natural forest had significantly the highest species richness (zero-inflated negative binomial regression $\chi^2 = 81.36, P < 0.001$) with an average of four to five species per plot as compared to tree plantations which ranged from one to three species per plot only. Of the exotic tree plantations, only teak had statistically similar ($P = 0.1143$) species richness with the natural forest and supported a relatively higher number of species when compared to mahogany and mangium plantations. Similarly, the diversity of native regeneration was significantly higher (Welch’s ANOVA $F_{[3,36]} = 11.42, P < 0.001$) in the natural forest ($H = 1.79 \pm 0.12$) than in the tree plantations. Teak stand was again the closest to natural forest among the tree plantations with diversity index of 0.87 ($\pm 0.09$), although this value is not significantly different from those in the mangium stand ($H = 0.79 \pm 0.22$). The latter also shares similarity in native species diversity with the mahogany stand ($H = 0.32 \pm 0.11$), which had the lowest diversity index among the sites. The use of ENS further confirmed the differences in species diversity consistent with the result of $H$ indices. The ENS result further showed that natural forest was significantly more diverse among the stands (Welch’s ANOVA $F_{[3,36]} = 5.27, P < 0.05$) except, however, with the ENS result among exotic plantations which indicated no significant difference. The natural forest had a true diversity of $6.33 (\pm 0.63)$ effective number of species, which is three to six times higher than mahogany (ENS = 1.44 ± 0.15), mangium (ENS = 2.74 ± 0.67), and teak stand (ENS = 2.47 ± 0.21), respectively.

Moreover, the steep curves in the rarefaction plot indicated that there is still a small fraction of species, particularly in the mangium, teak, and natural forest, that remain to be accounted. These are expected to yield up to 10, 20, and 30 more species, respectively, if the number of sample plots is increased (Figure 8). In mahogany, the curve slope is flatter to the right, suggesting that a reasonable number of individual samples of different species have been taken; thus, additional sampling plots are likely to yield only a few additional species.

### Discussion

In a country like the Philippines, where agriculture and forest resources are the primary and often the only viable sources of income, addressing deforestation and consequently native species extinction would remain a huge challenge unless a well-informed ecologically and socially sustainable restoration and forest management framework is identified and developed.

The country has a long history of using fast-growing exotic tree species in its effort to abate land degradation and recreate immediate forest cover in denuded forestlands, which account for at least 70% of the country’s total land area. Although some species such as mahogany, gmelina, teak, mangium, and falcata have been proven to thrive well, becoming a valuable resource for the country’s pulp and/or timber needs; past reforestation efforts were generally considered a failure as issues of high mortality in most areas due to poor site adaptation and occurrence of pests and diseases were a major drawback of the program. And, just recently, in the country’s biggest reforestation effort (dubbed as the ‘national greening program’), the national government has shifted to using both native and exotic species (mostly mahogany and gmelina) to reforest 1.5 million ha nationwide by 2016. But, as with previous reforestation efforts, the results from the latest assessment are poor, although the reason this time was largely due to inefficient and/or ineffective implementation and monitoring mechanisms. While the recent forest rehabilitation design is plausible, as it has the potential to address both the ecological and social needs of the country, the involvement of exotic species in this massive rehabilitation effort has again attracted strong criticism and opposition from several environmental groups, mainly because of its negative implications for biodiversity conservation.

The findings of this study, however, have indicated some potential for exotic tree plantations to support regeneration of some important indigenous species, which can be expected to colonize these areas given the higher density of native saplings in the plantations as compared to natural forest. Sapling...
density was significantly higher in the exotic stands than in natural forest, particularly in the mangium and mahogany plantations. Ruiz-Jaén and Aide (2005) noted that colonization of woody saplings and/or seedlings are, along with diversity and ecological processes, one of the measures of restoration success. The seemingly sparse number of native saplings in the natural stand as compared to seedlings could be due to high competition for available resources in the natural stand; or a particular case of seed limitation for the low seedling densities in the case of tree plantations (Gonzales and Nakashizuka 2010).

However, given the proximities of the stands in the study, where animals like birds can disperse them easily, it is more likely that the germination of dispersed seeds was limited by the extensive growth of grasses in the tree plantations, particularly Imperata cylindrica as a result of previous timber harvesting activities in the area. According to French (2010) even small scale disturbances can provide opportunities for weed establishment, including chances for invasion of the habitat over time. Imperata cylindrica is known to colonize open niches and can limit forest seedlings’ establishment leading to poor representation of native species regeneration (Kiaporanis and Nimiago 2005) as seedlings are highly sensitive to competition, especially during initial growth (Pardos et al. 2005). This scenario is therefore more of a management concern rather than a negative influence associated with exotic tree plantations.

In terms of species richness, the present study recorded a total of 46 native tree species from 25 families in which almost half of this number of species was found under the three exotic tree plantations. As expected, natural forest had the highest number of species (28) owing to its diverse overstorey cover (Kawekrom et al. 2005; Omor and Luukkanen 2011), and this was followed by teak (15 species), mangium (11 species), and mahogany with only three species. The better species regeneration in the natural forest could be an indication of more favorable growing conditions as the tree plantation sites in the study were formerly disturbed and/or logged-over areas, although the influence of the sites’ microenvironment needs further investigation.

Each stand exhibited uniqueness in species composition, with only 5%–15.91% similarities observed despite the stands’ proximity, particularly natural forest, mangium, and teak stands. Zanne and Chapman (2001), however, observed that not all sites adjacent to a seed source had equal amount of regeneration (i.e. species richness and stem density) as variability in site characteristics (e.g. soil attributes, percent herb or grass/fern cover) exists even among similar vegetation type at the same sites. The effect of distance from seed sources was more likely a key limitation for mahogany plantation being the farthest or isolated from natural forest, which may explain its very low native species regeneration as compared to other exotic plantations. In fact, no similarities in species composition between mahogany and natural forest were noted in the study. Distance from seed sources has been identified as one of the most significant factors influencing tree species colonization in different successional sites such as plantations (e.g. Parrotta, Knowles, et al. 1997; Hérault et al. 2004; Koonkhunthod et al. 2007; Gonzales and Nakashizuka 2010; Nagaïe et al. 2012), where a significant decrease in species richness with increasing distance to the source was a common observation.

Nonetheless, the observed differences in the dominant species at each stand with the natural forest being dominated by K. glomerata, C. brevistyla, and L. domesticum; the mangium stand by M. compressum var. inodorum, M. multiglandulosa, and S. contorta; the teak stand by S. philippinensis, K. glomerata, and P. indica forma echinatus; and the mahogany stand by S. contorta, A. odoratissimus, and C. cainito may also suggest that different species may have different habitat preference. This varying preference among species only proves that natural regeneration is an unpredictable process (Pardos et al. 2005), which may need long-term monitoring to understand the dynamics and patterns of natural regeneration over time. The study noted that the majority of species (96%) found in the four stands can be dispersed by animals, indicating the potential benefits of these exotic tree stands as shelter or nesting sites, which are necessary to facilitate seed dispersal – a critical component in the natural process of regeneration and spread of native vegetation.

The Shannon-Wiener indices (H), on the other hand, range from 0.32 (three species) to 1.79 (28 species) with the lowest and highest diversity values being observed, respectively, in mahogany plantation and natural forest. Shannon-Wiener indices ranges are typically from 1.5 to 3.5 and rarely reach 4.5, with a higher index value meaning higher species diversity (Ponce-Hernandez et al. 2004). Natural forest, as expected, was the most favorable habitat and was generally three to six times more diverse and species rich than the exotic tree plantations, followed by teak stand, mangium, and mahogany stands. Interestingly, the occurrence of native species in the tree plantations that include some of the threatened species such as the critically endangered S. contorta, P. indica forma echinatus, and endangered V. parviflora suggests that, when all ecological factors at play are equal, exotic tree plantations, especially teak and mangium, can be as favorable habitat as the native forest for biodiversity conservation. In particular, teak and mangium plantations have shown potential to support significant native species regeneration despite being considered a poor choice of habitat for native forest species (Healey and Gara 2003). However, the number of species recorded under the teak and mangium plantations in the present study was way below that of a teak stand in Thailand with 37 species (Koonkhunthod et al. 2007) or a mangium stand in Indonesia with 89 tree species (Kuusipalo et al. 1995); although this kind of comparison may be influenced by factors such as the age of the plantation, climate, and sampling procedure used (Koonkhunthod et al. 2007).

Moreover, while long distance from the natural forest could be a limiting factor for the low native species regeneration (i.e. diversity and richness) in the mahogany plantation, the results suggest that mahogany could be a poor choice as a tool to facilitate native regeneration when compared to teak and mangium stands. Mahogany has been observed in the Philippines by Baguinon et al. (2005) to be bio-invasive, which could limit native regeneration; and hence can be a cause of concern for native conservation as it may compete with native species for the various resources available. However, further investigation is strongly recommended to ascertain the role or influence of mahogany plantations on native tree species regeneration, as other studies in Fiji observed abundant recruitment of native tree species (totaling up to 69 species from 34 families) under mahogany stand (Tuiwawa and Keppel 2013). Lugo (1992) also noted significant species
richness of native tree species regeneration in the understory of a 50-year-old mahogany plantation in Puerto Rico, comparable to that of a second growth forest.

Nonetheless, the findings of this study are a departure from the generalization that monoculture tree plantations, especially those involving exotic species, cannot support native species regeneration. Our study showed some potential of the exotic tree plantations to facilitate native regeneration, which, if properly managed (e.g. continued protection, application of weed/grass control, consideration of their proximity to seed sources), could be expected to develop into a native species dominated forest. Although needing further investigation, our results suggest that distance from natural forest or possible seed sources may be critical for the successful establishment and restoration of native species in exotic tree stands. Furthermore, more research is clearly required in deciding if exotic tree plantations can be a valuable component in the country’s indigenous forest rehabilitation and conservation programs. For instance, there is a need for long-term monitoring of indigenous forest recovery under tree plantations including the kind of management intervention needed to improve the natural succession trajectory over time. Because exotic species can be invasive, it would be critical also to investigate possible management complications and ecological impacts of their use in forest restoration programs, including those that are, or will be, used in commercial plantations.

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Disclosure statement

No potential conflict of interest was reported by the authors.

ORCID

Adrian M. Tulod http://orcid.org/0000-0001-9704-0567

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