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Choosing species for reforestation in diverse forest communities: social preference versus ecological suitability

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Abstract. Choosing species for reforestation programs or community forestry in species-rich tropical rainforest ecosystems is a complex task. Reforestation objectives, social preferences, and ecological attributes must be balanced to achieve landscape restoration, timber production, or community forestry objectives. Here we develop a method to make better species choices for reforestation programs with native species when limited silvicultural information is available. We conducted community surveys to determine social preference of tree species and inferred their ecological suitability for open-field plantations from growth rates and frequency in forest plots at different successional stages. Several species, for which silvicultural data was available, were correctly classified as promising or unsuitable for open-field reforestation. Notably, we found a strong negative correlation between ecological suitability indicators and socioeconomic preference ranks. Only a single outlier species ranked very high in both categories. This result highlights the difficulty of finding suitable native species for community forestry and offers an explanation why reforestation efforts with native species often fail. We concluded that the approach should be a useful first screening of species-rich forest communities for potential reforestation species. Our results also support the view that species-rich tropical rainforests are not an easily renewable natural resource in a sense that secondary forests will not provide an equivalent resource value to local communities.

Key words: life history traits; natural forests; Philippines forestry; reforestation with native species; social forestry; socioeconomic importance; tropical forests.

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INTRODUCTION

Government reforestation and community forestry programs are important activities to offset the results of many decades of deforestation, which has left many countries and jurisdictions with an impoverished and degraded land base (Hansen et al. 2013). Tropical forest ecosystems have been particularly affected, and still experience a poor ratio of forest loss to forest gain (3.6:1 for the 2000–2010 period), despite promoting natural recovery after forest harvest and afforestation and reforestation efforts (Hansen et al. 2013). Loss of tropical forest cover by geographic regions is most severe in Southeast Asia (37% of total area remaining in 2005), followed by Central America and Central Africa with 45%, and the Amazon basin with 56% forest cover remaining (FAO 2005). Primary forest in Southeast Asia and Central America are most affected with 16% and 18% original forest cover remaining intact, respectively (FAO 2005). Deforestation rates also remain unsustainable with 1.2%/yr of net forest loss between 1990 and 2005 in Central America, 0.7%/yr net forest loss in Southeast Asia, and 0.5%/yr in Central Africa and the Amazon region (FAO 2005).

Within major geographic regions, countries with high accessibility to forests and high population densities have little primary forest left, often limited to mountainous regions. For example, in Southeast Asia, the Philippines, Vietnam, Laos, and Cambodia had less than 6% primary forest cover left in 2005 (FAO 2005). In these countries, land degradation, erosion, water supply concerns, and landslides have often become common problems (Sidle et al. 2006). In the recent decades, the occurrence of landslides in Southeast Asia increased by five times from 1970 levels due to deforestation (Forbes and Broadhead 2013). With tropical cyclones making landfall five to seven times a year in the Philippines, disasters are common with 35.4 per million fatalities reported between 1950 and 2009, the second highest rate worldwide (Forbes and Broadhead 2013).

In response, government programs aimed at reforestation and protection of remaining forests were created. Many countries in Southeast Asia imposed logging bans on natural forest lands, phased out large timber operations and created large-scale government reforestation programs and community forestry programs (Tumaneng-Diete et al. 2005, McElwee 2009, Pulhin and Dressler 2009, Yonariza and Singzon 2010). Between 1985 and 1997 the percent of total protected areas increased from 4% to 9% worldwide (Zimmerer et al. 2004), and community forestry systems were widely adopted to produce locally required resources and reduce pressures on harvesting the remaining natural forests (Molnar et al. 2007). Community forestry programs worldwide increased by 70% to approximately 250 million hectares between 1985 and 2008 (Molnar et al. 2007). In addition, large-scale government reforestation program were created in many jurisdictions, including the Philippines, where government program reforested 560,000 hectares between 1990 and 2000 (Cruz et al. 2001).

Although large-scale reforestation programs that aim to rehabilitate degraded forest lands and improve watershed properties are laudable, one potential concern is that most of these initiatives rely on exotic species (Tolentino 2008). Large-scale reforestation programs in Southeast Asia primarily plant *Gemelina arborea*, pine, *Swietenia* sp., *Eucalyptus* sp., *Tectona grandis* and *Acacia* sp., because of readily available seed from international seed banks and well established nursery protocols and silvicultural treatments for these species. As a result, exotics amount to 80% of all trees planted in tropical reforestation efforts (Cruz et al. 2001, Tolentino 2008, Yonariza and Singzon 2010). Exotic species monocultures, however, may change natural habitat conditions, water balances, and nutrient cycles, typically in ways that do not support biodiversity of native species (Hooper et al. 2002). Additionally, exotic species can sometimes be vulnerable to large outbreaks of insect pests and diseases, such as the Dothistroma Needle Blight in Central America and Asia (Watt et al. 2009) and the Rhizina root disease in Southern Africa that have set back reforestation efforts to the beginning (Wingfield et al. 2001).

For large-scale reforestation and landscape restoration projects, native trees are therefore preferred (FAO 2014), although native tree planting is not easy to carry out for lack of silvicultural knowledge and regeneration ecology (Bautista 1990). Planting native species exclusively is not required for community reforestation projects with limited ecological impacts because of their small scale, but native species could also be beneficial here because the species may be well known and valued by local communities for their non-timber benefits (Peters et al. 1989, Lacuna-Richman 2002, Mangaoang and Pasa 2003, McElwee 2008). In the Philippines, community forestry with native species has shown some success in the form of "rainforestation farming" programs, which combine native species for reforestation with some exotic fruit trees to support local livelihoods (Goltenboth and Hutter 2004, Vilei 2009, Schneider et al. 2014). Similar initiatives are sponsored throughout the world by regional governments and international agencies, including the Reducing Emissions from Deforestation and forest Degradation (REDD+) program, the largest global initiative to combat deforestation, foster sustainable forest management, and fight rural poverty (Agrawal and Angelsen 2009).

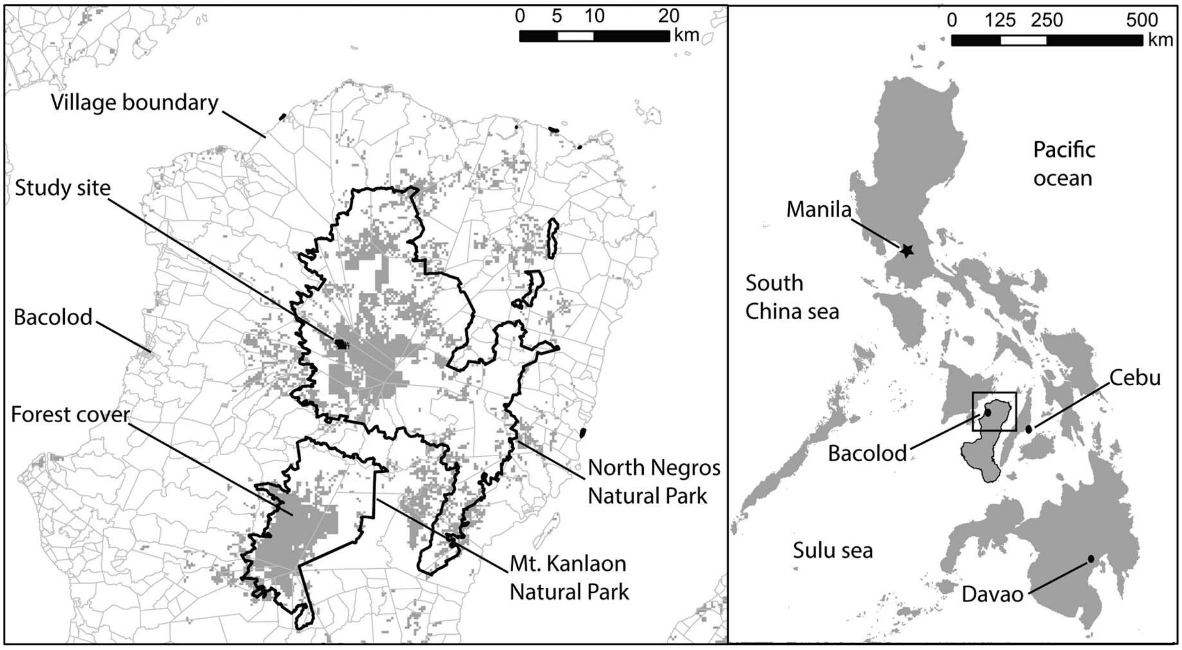

Fig. 1. Map of the study area located within the North Negros Natural Park on Negros Island in the Philippines. The questionnaires were collected from local stakeholders that live in the communities around the park.

In support of such programs, this research aims to make better-informed species selections for reforestation programs based on the ecology and socioeconomic desirability of tree species. The primary objective of this paper is to develop a screening method for potential reforestation species when silvicultural information is lacking. To identify early-succession species, suitable for reforestation under open field conditions, we investigate if standard inventory plot data can yield the required ecological information. Our expectation is that high frequency and basal area in early-successional plots is associated with pioneer species that will do well when planted in degraded sites. Our second objective is to assess social preferences and find species with early-successional ecological characteristics that are also highly valued as reforestation species by local communities. While our research is a regional case study, we aim at developing a simple, low-cost approach to screen species-rich communities for potential reforestation species that can be applied to other regions and forest communities.

METHODS

Study site and ecological data

Our study site is the North Negros Natural Park (NNNP), located in Silay City of Negros Occidental Province, Philippines (Fig. 1). The NNNP was initially established in 1935 as a timber reserve (Hamann et al. 1999, Hamann 2002), but expanded in 2005 and declared a Nature Reserve to protect high biodiversity, including many rare, endemic and endangered plant and animal species. Previous botanical assessment in 1999 found 92 different species of 39 families, with 50% of the trees coming from the *Dipterocarpaceae*, *Lauraceae*, *Sapotaceae*, *Burseraceae*, and *Melastomaceae* families. The most common individuals were of the *Litsea luzonica*, *Canarium asperum*, *Platea excels*, and *Palaquium* sp. species representing a sub-montane forest ecosystem without a pronounced dry season (Hamann et al. 1999).

The experimental study plots are on the northwestern slope of Mt. Mandalagan close to Mt. Silay (10°38.0′ N, 123°13.0′ E) at an average elevation of a 1000 m. The three experimental plots were chosen to represent early-, mid-, and late-successional stages. They include a 1.61-ha plot in an early-successional forest area cleared for pasture and abandoned in 1975, a 1.71-ha plot in a mid-successional forest destroyed through bombing during World War II and a 5.15-ha area of old-growth forest not known to be cleared in the past 100 years. In the entire forested area truly undisturbed forest is difficult to find because of proximity to settlement. However, areas surveyed are on a rugged terrain and prevented logging industry from harvesting trees. Low intensity use by the community involves rattan, firewood and seedling collection. Some areas of the forest have been sub planted with coffee by the community.

The plot size was determined via species-area curves, so that further increases in plot size would not capture additional species (except very rare species). While this approach captures most species that would be considered for reforestation in a local area, the approach does not yield comprehensive information for a larger geographic region, for example other islands in the Philippines. Rather, our approach is meant to be easily repeatable to screen other species-rich communities for local reforestation programs elsewhere.

Tree measurements were made three times over an 18 year period in 1995, 2003 and 2012 at all three successional plots, where all trees more than 10 centimeters DBH were measured and permanently tagged. Measurements for these trees included DBH, crown position and survival. New recruiting trees recorded during the 2003 and 2012 measurements were also permanently tagged. Seed sizes used in the analysis of this paper were obtained from a previous botanical survey (Hamann et al. 1999). Species diversity was evaluated based on frequency data using the Shannon–Weaver index (Shannon and Weaver 1948).

Survey of social preferences

A survey of local knowledge of native tree species in North Negros Natural Park was conducted through questionnaires and picture guide developed from the botanical inventory effort. The questionnaire was designed with open ended questions about forest use and given to community members employed as forest guards of the protected area. Forest guards were a useful focus group because they were recruited from the portion of the population that was forest-dependent, in part to provide alternative livelihoods to forest extraction. While they may not represent a completely unbiased sample of the preferences of the local forest-dependent population, they provided an opportunity to efficiently assemble a local knowledge base of social preferences and uses of local tree species. The questionnaires were given at the general assembly and included forest guards from five municipalities around the North Negros Natural Park, Silay, Talisay, Victorias, E.B. Magalona and Sagay with 47 respondents.

In compliance with Canadian research ethics requirements for community surveys, the involvement in the questionnaire was voluntary and the reasons and use of this data were explained to the respondents before giving the questionnaire. Participation could be terminated by the respondents at any time during or after the completion of the questionnaire and the incomplete questionnaires were not used in the analysis. Respondents had the right to request that their answers not be used in the study for a month after the completion of the questionnaire.

The questionnaire asked respondents to list tree species of importance to the community as a source of food, lumber, firewood, medicine, and charcoal making or to the forest for general ecological observations (for example, use of fruit trees by wildlife). The responded listed local names of tree species they considered important to the community with their uses. Local names were later matched with a scientific name using a species guide developed during the first forest assessment that contains 92 species of trees found in the North Negros Natural Park in 1995 (Hamann et al. 1999). The importance was ranked by combining the number of uses noted by respondents and the number of respondents that mention the species. The highest-ranking trees are those that have several uses and were mentioned by many people as important to the community.

Statistical analysis of plot data

To analyze which native species have an early-successional life history syndrome, we derived

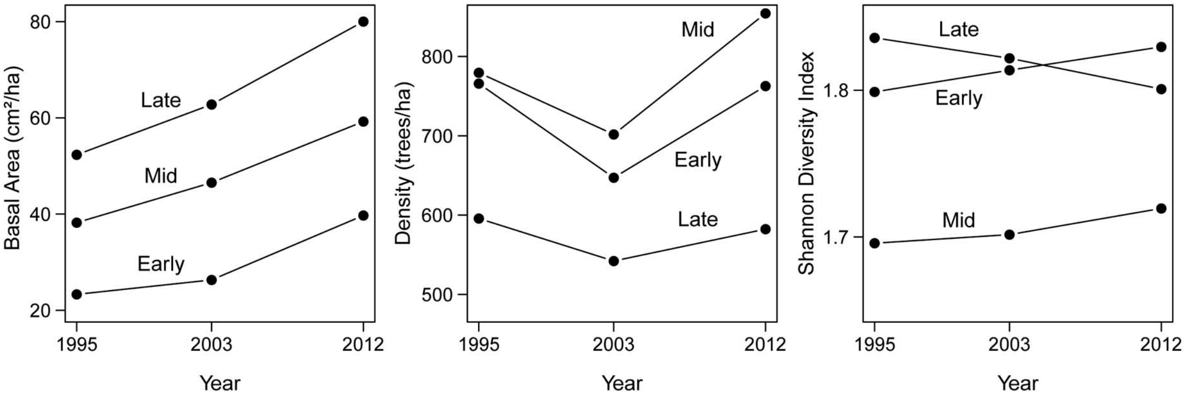

Fig. 2. Tree size, density and diversity of trees with a diameter at breast height larger than 10 cm from the early, mid and late-successional study plots in the North Negros Natural Park. The graph shows the change in basal area, density and species diversity over a period of 18 years from three surveys.

three variables for each species at each successional plot. They are: (1) relative frequency, calculated as the number of trees per hectare for each species, (2) basal area, calculated as the total stem area per hectare of each species, and (3) growth increment among measurements in 1995, 2003, and 2012, calculated as the relative change in basal area. Growth rates were missing for some species because measurements were not available in one of the survey times. Growth data was not always available in one interval for rare species because of mortality, or became available in the second interval because of recruitment. To use this data, we estimated the growth rate for the missing year with a linear mixed model implemented with the *ASReml* (Butler et al. 2007) for R programming environment (R Core Team 2014). Rather than calculating means across multiple survey times, we use best linear unbiased estimates (BLUEs) from the linear mixed model. These estimates conform to straight means if there are no missing values, and otherwise provides an unbiased estimate of the true mean.

To visualize groups of putative early- and late-successional life history syndromes, we applied nonmetric multidimensional scaling (NMDS) of the nine variables, implemented with *nmds* function of the *ecodist* package (Goslee and Urban 2007) for the R programming environment. This scaling technique works by calculating a multivariate distance matrix based on the available measurements, and subsequently ordinate the observations based on this matrix (Kruskal 1964). Here, we used a Bray-Curtis distance matrix, suitable for presence absence and quantitative measurements in community ecology data to ordinates the observations with NMDS. One purpose of the NMDS ordination is to search for similarities in the multivariate dataset to find easy to measure variables, such as frequencies, that may stand for other, harder to get variables such as growth rates. The separation of data by successional syndrome was verified by plotting seed sizes of known species, with larger seed sizes usually belonging to species of late-successional syndromes (Budowski 1965, Bazzaz and Pickett 1980). The species with missing data estimated with best linear unbiased predictions with *ASReml* were left out of the original ordination and their NMDS scores were later determined with the *predict* function to avoid a biased ordination because of poor data quality. The NMDS scores of species corresponding to successional syndromes were plotted against social preference data from the questionnaire, and the relationship was quantified with a Pearson correlation analysis.

RESULTS

Characteristics of experimental plots

The selected forest plots show largely typical characteristics of a successional series regarding basal area, tree density, and species diversity (Fig. 2). Notable is the reversal of expected

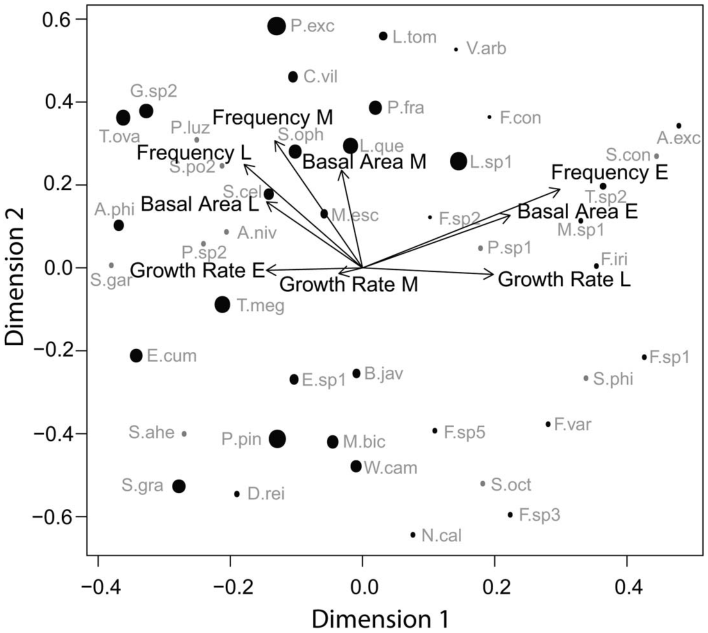

Fig. 3. Ordination of tree species by similarity in frequency and growth rate in early (E), mid (M) and late-successional (L) study plots. The size of dots represents seed size (which was not used in the ordination), except when data was not available (gray dots). Dimension 1 of the non-metric multidimensional scaling (NMDS) procedure separates early-successional species on the right from late-successional species on the left. Dimension 2 further separates putatively mid-successional species (high values) from late-successional species.

density in the early- and mid-successional plot, because of many trees in the early-successional plot having several stems counted as a single tree. Before the second field season, a strong typhoon hit the area and many smaller trees fell or were destroyed by branches falling from large trees, causing the density of small trees (DBH 10–20 cm) to decline. The basal area measurement, driven by large trees was not significantly reduced, however, with little mortality in the large diameter classes. Species diversity, measured by the Shannon–Weaver index, declined over time in the mature plot, while it increased in the successional plots. The early-successional plot's high diversity was driven by the evenness and not the richness of species. The early-successional plot has the fewest species but nearly half occur in high frequency (relative frequency > 5 trees/ha) compared with a third of species occurring in high frequency in mid- and late-successional plots.

Successional syndromes of tree species

The NMDS of ecological tree data (Fig. 3) explains 89% of the variance in the dataset in two dimensions. In NMDS, dimensions are not orthogonal, and therefore we do not report variance explained for individual dimensions. Vectors in the plot indicate how original measurements are related to NMDS dimension scores. The length reflects the strength of the correlation and the direction indicates positive or negative correlations between dimensions and original variables. Points in the plot represent tree species and their positions indicate associations with original measurements. For example, species toward the right have high frequency and basal area in the early-successional plot, and show high growth rates in the late-successional plot. Further, we scale the plots based on a life history trait often associated with a successional syndrome: seed size. For gray dots, no data on seed size was available.

Dimension 1 in Fig. 3 is strongly associated with an early-successional syndrome. High frequency and basal area on the early-successional plot are positively correlated with this axis, although the growth rate is negatively

Table 1. Local uses of native tree species in the North Negros Natural Park. The table summarizes selected data from the questionnaires with 47 forest guards that listed species of socioeconomic importance and their uses. The uses are grouped into lumber/construction (L), varnish/resin (V), charcoal/firewood (C), edible products (F), ecological-food for animals (E), medicine (M). The summary includes sum of total uses for each species (T), number of respondents that mention this species (N) and rank (R) that combines the number of uses and the number of respondents. Local names could refer to several related species (reflected by multiple codes), although only one primary species name is given. Refer to Appendix: Table A1 for a full list of local species uses.

		Uses						Statistics		
Species (Code)	Family	L	V	C	F	E	M	T	N	R
Agathis philippinensis (A.phi)	Araucariaceae	2	3	1	0	0	0	6	37	43
Palaquium luzoniensis (P.luz)	Sapotaceae	4	0	0	1	0	0	5	20	25
Shorea contorta (S.con)	Dipterocarpaceae	4	0	1	0	0	1	6	17	23
Garcinia brevirostris (G.bre/sp2)	Guttiferae	4	0	0	0	1	0	5	15	20
Syzygium gracile (S.gra/S.sp2)	Myrtaceae	4	0	0	0	1	0	5	15	20
Cinnamomum mercadoi (C.mer)	Lauraceae	3	0	2	1	0	1	7	12	19
Platea excelsa (P.exe)	Icainaceae	0	0	0	1	3	1	5	11	16
Shorea polysperma (S.pol)	Dipterocarpaceae	3	0	0	0	0	0	3	12	15
Dillenia philippinesis (D.phi)	Dilleniaceae	0	0	0	1	1	0	2	12	14
Prunus fragrans (P.fra)	Rosaceae	2	0	0	1	2	0	5	9	14
Lithocarpus sp. (L.sp1)	Fagaceae	1	0	0	1	2	1	5	8	13
Memexylon brachybotris (M.bra)	Melastomataceae	2	0	0	0	1	0	3	9	12
Mallotus sp.(M.sp1)	Melastomataceae	2	0	0	0	1	0	3	9	12
Bischofia javanica (B.jav)	Euphorbiaceae	2	0	0	0	1	0	3	8	11
Garcinia binucao (G.bin/sp1)	Guttiferae	0	0	1	1	0	0	2	8	10
Arthrocarpus heterophyllus(A.het)	Moraceae	1	0	0	1	0	0	2	7	9
Actinodaphne sp. (A.sp1)	Lauraceae	0	0	2	0	1	0	3	6	9
Ficus crysolepis (F.chr/sp6/sp8)	Moraceae	0	0	0	0	3	0	3	6	9
Myrica esculenta (M.esc)	Myricaceae	0	0	0	0	1	0	1	8	9
Shorea almon (S.alm)	Dipterocarpaceae	3	0	1	0	0	0	4	5	9
Pometia pinnata (P.pin)	Sapindaceae	1	0	1	1	0	1	4	4	8
Trema orientalis (T.ori)	Urticaceae	1	0	0	0	1	0	2	5	7
Memexylon cumingii (M.cum)	Melastomataceae	0	0	1	0	0	0	1	6	7
Alphitonia excelsa (A.exc)	Rhamnaceae	1	1	0	0	0	0	2	3	5
Palaquium sp. (P.sp1)	Sapotaceae	1	0	0	0	0	0	1	4	5
Ficus benjamina (F.ben/F.sp4)	Moraceae	0	0	0	0	1	0	1	3	4
Litsea tomentosa (L.tom)	Lauraceae	1	0	0	0	0	0	1	3	4
Ficus septica (F.sep)	Moraceae	0	0	0	0	1	0	1	2	3
Canarium villosum (C.vil)	Burseraceae	1	0	0	0	0	0	1	1	2

correlated, indicating that the species that fall on the right side of this plot have reached maturity on the early-successional plot. These species are also correlated with high growth relates on the late-successional plot, where this species group vigorously recruits in gaps after typhoon disturbances shortly after the plot series was established. Dimension 2 weakly separates late- and mid-successional species groups, with putative mid-successional species correlating primarily along the second dimension (species toward the top of the plot) and show little association with the first dimension. Putatively late-successional trees correlate with each dimension and are located toward the top and left areas of the plot. Large seed sizes are usually associated with mid- or late-successional species (Budowski 1965, Bazzaz and Pickett 1980) and are grouping on the left and top area of the ordination as well.

Socioeconomic importance of tree species

Surveys of social importance resulted in 71 tree species mentioned by survey participants, of which three-quarters were native and one-quarter introduced species (Appendix: Table A1). Of the native species, 35 were found in the ecological monitoring plots evaluated in this study (Table 1). Plant families that ranked high in social importance include *Dipterocarpaceae*, *Fabaceae* and *Lauraceae*, with five or more highly valued species each. *Dipterocarpaceae*, also known as Philippine mahogany are some of the most important tropical timber species. Dipterocarps

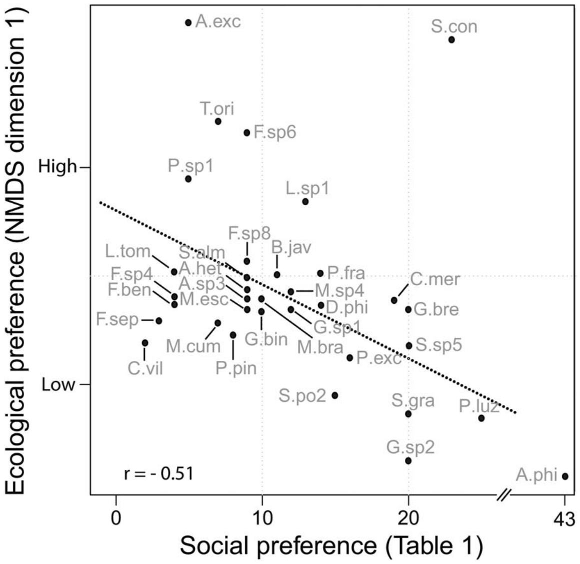

Fig. 4. Scatter plot of native species ranked by socioeconomic importance (x-axis) and inferred ecological suitability for open field plantation (y-axis). The trend line and r-value exclude the outliers (A.phi and S.con). However, the relationship remains statistically significant and moderately strong even with the outliers included ($r = -0.41$).

are primarily used locally for construction and furniture making. *Fabaceae* contain fast growing, nitrogen fixing tree species with dense wood valued for charcoal production and lumber for construction. *Lauracea* are the second dominant tree species group in the upland forests of Southeast Asia, highly valued for their timber as well.

The most common uses mentioned across all species are traditional timber values. Construction, furniture making, and lumber constituted 48% of all mentioned use of native and non-native trees by the community in this study (Appendix: Table A1). Next in importance are the species used for charcoal and firewood, constituting 17% of all species-use combinations. Species that had mostly non-timber uses accounted for 28% of species-use combinations, with *Fabaceae*, *Guttiferae*, *Icainaceae*, *Moraceae*, *Lauraceae* and *Sapotaceae* families mentioned prominently for their use as fruit trees for wildlife and for human consumption. Medicinal use accounted for 5.6%, and included species of the families *Moraceae*, *Meliaceae*, *Lauraceae*, *Fagaceae*, and *Icainaceae*.

To identify species for reforestation that have economic uses and are suitable for open-field reforestation, we plotted the Dimension 1 from Fig. 3 where high scores indicated early-successional species as a function of their social preference from Table 1 (Fig. 4). The most desirable species should fall into the upper right quarter of this graph. Notably, we observed a negative correlation between inferred ecological suitability and social preference ranks ($r = -0.41$, $p = 0.015$, df = 42). If two outliers are excluded (Fig. 4; A.phi – *Agathis philippinensis* and S.con – *Shorea contorta*), the correlation is more pronounced ($r = -0.51$, $p = 0.003$, df = 42).

DISCUSSION

Social preference versus ecological suitability

The negative association between ecological suitability indicators and social preference ranks highlights the difficulty of finding suitable native species for community forestry where social acceptability is essential and planting efforts are often restricted to abandoned farmland. Nevertheless, the realism of the proposed approach is highlighted particularly saliently by the two outlier species in Fig. 3, *Agathis philippinensis* and *Shorea contorta*.

Agathis philippinensis has a native range from Indonesia to the Philippines and related species are well regarded in Australia and New Zealand, where the species group is known as kauri, valued for a variety of products made from their resin (Lacuna-Richman 2006). *Agathis philippinensis* cannot survive in open-field plantations but can only be sub planted in secondary forest in enrichment efforts (Orwa et al. 2009). Our analysis correctly identified the species as late-successional species. *Shorea contorta* is a dipterocarp species native to Southeast Asian forests, known as white lauan in the Philippines and is marketed for its wood qualities worldwide. In this study, it had a high preference rank as well (Table 1; Appendix: Table A1). In contrast with *Agathis* this species is known to grow well in full sunlight unlike most other dipterocarp species (Schneider et al. 2014) and was correctly classified in this study as an early-successional species.

Further, we note that there are many early-successional species in the category of medicinal uses. Thus, in this use category, the negative relationship between socioeconomic preference and ecological suitability is weaker and provides an opportunity for species selection. Some examples include the use of *Ficus septica* roots to cure boils and to eliminate access water, while the leaves have antirheumatic properties and relieve headaches (Lanting and Palaypayon 2002). The bark of *Shorea contorta* is used for cough or as an astringent while the wood decoction is used to treat tumors (Lanting and Palaypayon 2002). Another example is *Casuarina equisetifolia*, the leaves of which have antifungal and antibacterial properties and are used traditionally to treat gonorrhea and stomach infections (Doyle and Aalbersberg 1998). Species with medicinal uses are commonly found in the early-successional group, which produce a host of secondary metabolites and allelochemicals as protection against herbivores and for other interspecies defenses (Bryant et al. 1983).

Native forests are not an easily renewable resource

Our results indicate that species-rich Dipterocarp forests may not be an easily renewable natural resource with respect to their value to forest-dependent communities. Late-successional tree species have the highest resource value, whereas, early-successional secondary forests and planted forests do not serve social needs of forest-dependent communities as original native forests. Previous research on economic valuation of mature tropical forest has shown that old-growth forest bring the highest economic benefits from non-timber forest products (Peters et al. 1989) and forest products from diverse forests can provide a diversity of marketable products to local communities (Pattanayak and Sills 2001). Alternative uses, such as plantation forestry or a cattle pasture compared unfavorably in valuations (Peters et al. 1989). Although this study found traditional timber uses rather than non-timber products to be the most important socioeconomic feature in contrast with Peters et al. (1989), those values were largely restricted to late-successional species. Our results therefore confirm the importance of forest conservation and restraint in resource extraction from existing natural forests.

Our data also points to a potential explanation of why reforestation efforts with native species often fail. Typically, highly valued native trees used in local reforestation programs are late-successional species. However, late-successional species usually cannot survive the environmental stress associated with open-field plantation such as high insolation, dry and degraded soil conditions (Uhl et al. 1988). In the Philippines, for example, large-scale reforestation programs use a few native species, mostly dipterocarps, planted on small areas that show lower growth rates compared with exotics (Otsamo et al. 1997, Tolentino 2008). Other programs in the Philippines that focus on reforestation with a high diversity of native species, such as the "rainforestation farming" program, are small in scale and expensive to implement (Vilei 2009).

In reforestation compromises may be necessary

The negative association between ecological suitability indicators and social preference ranks highlights the difficulty of finding suitable native species for community forestry where social acceptability is essential and planting efforts are often restricted to abandoned farmland. In our surveys we noted many replies for uses of exotic species in local communities, accounting for one-quarter of all species-use combinations. Including exotic species in small-scale, short rotation community forestry programs appears to be a sensible choice according to our surveys. Such programs should focus on high value and employ a variety of species for generating a reliable source of income (Lamb 1998, Leopold et al. 2001, Hartley 2002). The alternative choice of fast-growing, early-successional native species with few local uses may not meet the necessary socioeconomic criteria.

For large-scale afforestation projects to restore watershed services, and restore forest cover at the landscape level, we would discourage using exotic species. Although it is relatively easy to successfully establish large-scale monocultures of exotic trees with well-known propagation and planting techniques, the associated loss of biodiversity and the risk of unintended ecological impacts are well documented (Hooper et al. 2002, Sayer et al. 2004, Carnus et al. 2006). In this case, planting a variety of native early-successional species would be the best first step to reestablish forest cover (Chazdon 2008). Such secondary forests will not provide immediate benefits to the communities, but over time natural regeneration (Lee et al. 2005, Piiroinen et al. 2015, Wolfe et al. 2015) or enrichment planting with highly valued native species (Paquette et al. 2009, Ashton et al. 2014) may restore the original economic and ecological benefits to local communities. Alternatively, high-value late-successional species may be established more rapidly by the initial use of early-successional species as nurse trees to shade out grasses and herbs and planting high valued canopy species after a few years (Ashton et al. 2001).

General guidelines for species selection

Our original expectation of being able to identify a suite of easy to measure, quantitative indicators for promising reforestation species from observation plots proved more difficult than expected. For example, growth rates of early-successional species in our study were highly dependent on the exact stand history. In the early-successional plot used in this study early-successional species were already reaching maturity, resulting in low growth rates. In contrast, regeneration in forest gaps after typhoon damage in the late-successional plot resulted in rapid growth of early-successional species. Nevertheless, grouping species by similarity in their inventory plot measurements (Fig. 3), yielded interpretable ecological information and we would expect useful (while not necessarily identical) results if the approach was applied elsewhere in a different forest community with plots of somewhat different successional histories.

Information on suitability of species for reforestation under open-field conditions and degraded sites can, of course, most reliably be obtained from planting trials, but they involve an investment of time and resources if carried out for many species. Such trials can identify species with growth rates and survival that compares favorably with exotic species, as research in Costa Rica (Butterfield 1996), Panama (Hooper et al. 2002, Wishnie et al. 2007), and Indonesia (Otsamo et al. 1997). The suitability of a range of native species for reforestation can further be enhanced by an initial cycle of genetic tree improvement (Leakey and Simons 1997). However, this involves long-term investments that can only be made for a few species.

A first round of screening of species-rich forest communities for suitable reforestation species therefore remains an important problem to solve. Life history attributes, such as small seed size, also corresponded well to successional syndromes inferred from inventory plot statistics. Data may be obtained for other regions from databases such as TRY (http://www.try-db.org). Similarly, plot inventory data may be obtained for analysis from international data collection efforts such as BIEN (http://bien.nceas.ucsb.edu). We propose that screening for such indicators of early-successional life history strategies should be an important first step in narrowing species choices for reforestation. Ecological indicators, combined with social surveys data can rapidly be generated to support new, large-scale reforestation programs such as those initiated under the Reducing Emissions from Deforestation and Forest Degradation (REDD+) programs that include objectives such as on biodiversity conservation and providing livelihoods for the local communities (Agrawal and Angelsen 2009).

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LITERATURE CITED

Agrawal, A., and A. Angelsen. 2009. Using community forest management to achieve REDD+ goals. Pages 201–212 *in* A. Angelsen, editor. Realising REDD+. National strategy and policy options. Center for

International Forestry Research, Bogor Barat, Indonesia.

Ashton, M. S., C. V. S. Gunatilleke, I. A. U. N. Gunatilleke, B. M. P. Singhakumara, S. Gamage, T. Shibayama, and C. Tomimura. 2014. Restoration of rain forest beneath pine plantations: a relay floristic model with special application to tropical South Asia. Forest Ecology and Management 329:351–359.

Ashton, M. S., C. V. S. Gunatilleke, B. M. P. Singhakumara, and I. A. U. N. Gunatilleke. 2001. Restoration pathways for rain forest in southwest Sri Lanka: a review of concepts and models. Forest Ecology and Management 154:409–430.

Bautista, G. M. 1990. The forestry crisis in the Philippines: nature, causes, and issues. Developing Economies 28:67–94.

Bazzaz, F. A., and S. T. A. Pickett. 1980. Physiological ecology of tropical succession: a comparative review. Annual Review of Ecology and Systematics 11:287–310.

Bryant, J. P., F. S. Chapin, III, and D. R. Klein. 1983. Carbon/nutrient balance of boreal plants in relation to vertebrate herbivory. Oikos 40:357–368.

Budowski, G. 1965. Distribution of tropical American rain forest species in the light of successional processes. Turrialba 15:40–42.

Butler, D., B. R. Cullis, A. R. Gilmour, and B. J. Gogel. 2007. ASReml-R reference manual. Technical Report, Queensland Department of Primary Industries, Brisbane, Queensland, Australia.

Butterfield, R. P. 1996. Early species selection for tropical reforestation: a consideration of stability. Forest Ecology and Management 81:161–168.

Carnus, J. M., J. Parrotta, E. Brockerhoff, M. Arbez, H. Jactel, A. Kremer, D. Lamb, K. O'Hara, and B. Walters. 2006. Planted forests and biodiversity. Journal of Forestry 104:65–77.

Chazdon, R. L. 2008. Beyond deforestation: restoring forests and ecosystem services on degraded lands. Science 320:1458–1460.

Cruz, R. V. O., et al. 2001. State-of-the-art review of forest restoration in the Philippines. College of Forestry and Natural Resources, University of the Philippines Los Baños College, Laguna, Philippines.

Doyle, M., and W. Aalbersberg. 1998. Medicinal plants in the South Pacific. World Health Organization, Manila, Philippines.

FAO [Food and Agriculture Organization]. 2005. Global forest resources assessment 2005. Progress towards sustainable forest management. Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO [Food and Agriculture Organization]. 2014. The state of the world's forest genetic resources: thematic study. Genetic considerations in ecosystem restoration using native tree species. Food and Agriculture Organization of the United Nations, Rome, Italy.

Forbes, K., and J. Broadhead. 2013. Forests and landslides: the role of trees and forests in the prevention of landslides and rehabilitation of landslide-affected areas in Asia. Second edition. RAP, Food and Agriculture Organization of the United Nations, Rome, Italy.

Goltenboth, F., and C.-P. Hutter. 2004. New options for land rehabilitation and landscape ecology in Southeast Asia by "rainforestation farming." Journal for Nature Conservation 12:181–189.

Goslee, S. C., and D. L. Urban. 2007. The ecodist package for dissimilarity-based analysis of ecological data. Journal of Statistical Software 22:1–19.

Hamann, A. 2002. The North Negros Forest Reserve: a biodiversity hotspot at risk. Silliman Journal 43:83–90.

Hamann, A., E. B. Barbon, E. Curio, and D. A. Madulid. 1999. A botanical inventory of a submontane tropical rainforest on Negros Island, Philippines. Biodiversity and Conservation 8:1017–1031.

Hansen, M. C., et al. 2013. High-resolution global maps of 21st-century forest cover change. Science 342:850–853.

Hartley, M. J. 2002. Rationale and methods for conserving biodiversity in plantation forests. Forest Ecology and Management 155:81–95.

Hooper, E., R. Condit, and P. Legendre. 2002. Responses of 20 native species to reforestation strategies for abandoned farmland in Panama. Ecological Applications 12:1626–1641.

Kruskal, J. B. 1964. Nonmetric multidimensional-scaling: a numerical-method. Psychometrika 29:115–129.

Lacuna-Richman, C. 2002. The socioeconomic significance of subsistence non-wood forest products in Leyte, Philippines. Environmental Conservation 29:253–262.

Lacuna-Richman, C. 2006. The use of non-wood forest products by migrants in a new settlement: experiences of a Visayan community in Palawan, Philippines. Journal of Ethnobiology and Ethnomedicine 2:36.

Lamb, D. 1998. Large-scale ecological restoration of degraded tropical forest lands: the potential role of timber plantations. Restoration Ecology 6:271–279.

Lanting, M. V., and C. M. Palaypayon. 2002. Forest tree species with medicinal uses. Ecosystems Research and Development Bureau, Laguna, The Philippines.

Leakey, R. R. B., and A. J. Simons. 1997. The domestication and commercialization of indigenous trees in agroforestry for the alleviation of poverty. Agroforestry Systems 38:165–176.

Lee, E. W. S., B. C. H. Hau, and R. T. Corlett. 2005. Natural regeneration in exotic tree plantations in Hong Kong, China. Forest Ecology and Management 212:358–366.

Leopold, A. C., R. Andrus, A. Finkeldey, and D. Knowles. 2001. Attempting restoration of wet tropical forests in Costa Rica. Forest Ecology and Management 142:243–249.

Mangaoang, E. O., and A. E. Pasa. 2003. Preferred native tree species for smallholder forestry in Leyte. Annals of Tropical Research 25:25–30.

McElwee, P. 2009. Reforesting "bare hills" in Vietnam: social and environmental consequences of the 5 million hectare reforestation program. AMBIO: A Journal of the Human Environment 38:325–333.

McElwee, P. D. 2008. Forest environmental income in Vietnam: household socioeconomic factors influencing forest use. Environmental Conservation 35:147–159.

Molnar, A., M. Liddle, C. Bracer, A. Khare, A. White, and J. Bull. 2007. Community-based forest enterprises their status and potential in tropical countries. ITTO Technical Series (International Tropical Timber Organization), Food and Agriculture Organization of the United Nations, Rome, Italy.

Orwa, C., A. Mutua, R. Kindt, R. Jamnadass, and S. Anthony. 2009. Agroforestree database: a tree reference and selection guide. Version 4.0. World Agroforestry Centre, Nairobi, Kenya.

Otsamo, A., G. Ådjers, T. Hadi, J. Kuusipalo, and R. Vuokko. 1997. Evaluation of reforestation potential of 83 tree species planted on Imperata cylindrica dominated grassland: a case study from South Kalimantan, Indonesia. New Forests 14:127–143.

Paquette, A., J. Hawryshyn, A. V. Senikas, and C. Potvin. 2009. Enrichment planting in secondary forests: a promising clean development mechanism to increase terrestrial carbon sinks. Ecology and Society 14:31.

Pattanayak, S. K., and E. O. Sills. 2001. Do tropical forests provide natural insurance? The microeconomics of non-timber forest product collection in the Brazilian Amazon. Land Economics **77**:595–612.

Peters, C. M., A. H. Gentry, and R. O. Mendelsohn. 1989. Valuation of an Amazonian rainforest. Nature 339:655–656.

Piiroinen, T., P. Nyeko, and H. Roininen. 2015. Natural establishment of indigenous trees under planted nuclei: a study from a clear-felled pine plantation in an afrotropical rain forest. Forest Ecology and Management 345:21–28.

Pulhin, J. M., and W. H. Dressler. 2009. People, power and timber: the politics of community-based forest management. Journal of Environmental Management 91:206–214.

R Core Team. 2014. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Sayer, J., U. Chokkalingam, and J. Poulsen. 2004. The restoration of forest biodiversity and ecological values. Forest Ecology and Management 201:3–11.

Schneider, T., M. S. Ashton, F. Montagnini, and P. P. Milan. 2014. Growth performance of sixty tree species in smallholder reforestation trials on Leyte, Philippines. New Forests 45:83–96.

Shannon, C. E., and W. Weaver. 1948. A mathematical theory of communication. Bell System Technical Journal 27:379–423.

Sidle, R. C., A. D. Ziegler, J. N. Negishi, A. R. Nik, R. Siew, and F. Turkelboom. 2006. Erosion processes in steep terrain: truths, myths, and uncertainties related to forest management in Southeast Asia. Forest Ecology and Management 224:199–225.

Tolentino, E. L. 2008. Restoration of Philippine native forest by smallholder tree framers. Pages 319–342 *in* D. L. Snelder, editor. Smallholder tree growing for rural development and environmental services. Springer, Gainesville, Florida, USA.

Tumaneng-Diete, T., I. S. Ferguson, and D. MacLaren. 2005. Log export restrictions and trade policies in the Philippines: bane or blessing to sustainable forest management? Forest Policy and Economics **7**:187–198.

Uhl, C., R. Buschbacher, and E. A. S. Serrao. 1988. Abandoned pastures in Eastern Amazonia. I. Patterns of plant succession. Journal of Ecology 76:663–681.

Vilei, S. 2009. Adoptability of a complex agro-forestry project for smallholders on a Philippine island. Pages 292–305 *in* Small Scale Forestry Symposium. Division of Forestry and Natural Resources, Morgantown, West Virginia, USA.

Watt, M. S., D. J. Kriticos, S. Alcaraz, A. V. Brown, and A. Leriche. 2009. The hosts and potential geographic range of Dothistroma needle blight. Forest Ecology and Management 257:1505–1519.

Wingfield, M., J., B. Slippers, J. Roux, and B. Wingfield, D. 2001. Worldwide movement of exotic forest fungi, especially in the tropics and the southern hemisphere. BioScience 51:134–140.

Wishnie, M. H., D. H. Dent, E. Mariscal, J. Deago, N. Cedeño, D. Ibarra, R. Condit, and P. M. S. Ashton. 2007. Initial performance and reforestation potential of 24 tropical tree species planted across a precipitation gradient in the Republic of Panama. Forest Ecology and Management 243:39–49.

Wolfe, B., D. Dent, J. Deago, and M. Wishnie. 2015. Forest regeneration under Tectona grandis and Terminalia amazonia plantation stands managed for biodiversity conservation in western Panama. New Forests 46:157–165.

Yonariza, and S. B. Singzon. 2010. Post-logging ban timber tree planting in Thailand and the Philippines. Pages 1–28 *in* G. C. J. Saguiguit, editor.

Seventh SER European Conference on Restoration Ecology, 24–26 March 2010. Agriculture and Development Discussion Paper Series, Avignion, France.

Zimmerer, K. S., R. E. Galt, and M. V. Buck. 2004. Globalization and multi-spatial trends in the coverage of protected-area conservation (1980-2000). Ambio 33:520–529.

SUPPLEMENTAL MATERIAL

ECOLOGICAL ARCHIVES

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