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Full Length Research Paper

A study of five rare tree species in forests on limestone hills of northern Vietnam

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Rare tree species may become endangered as a result of forest conversion and disturbance. The availability of reliable ecological information on such tree species may help in conservation efforts, but such data is often limited. We studied the abundance of five red-listed tree species in an old-growth forest growing on limestone hills in northern Vietnam and identified potentially relevant ecological factors. Three of the studied species are largely restricted to limestone hills and regarded as specialists (*Excentrodendron tonkinense*, *Chukrasia tabularis* and *Garcinia fagraeoides*), while the two other species (*Parashorea chinensis* and *Melientha suavis*) are more widely distributed. The study species were recorded across 40 initial random plots at a range of frequencies (13 - 60%) and at relatively low stem densities (0.4 - 4% of the total stems). Using the adaptive cluster sampling method (ACS), we found all species to be relatively abundant across the site. The density of the three specialist species tended to increase from lower to upper slope positions, which was consistent with an increase in rock-outcrop cover, steepness of the slope, and a reduced soil depth. However, even on rocky, steep sites we found a high variation in tree densities of these species among plots, which could not be explained by the recorded ecological factors. The non-specialist species did not occur on mostly rocky sites. We regard these results as a potential baseline from which impacts of logging can be addressed and to inform potential management.

Key words: Adaptive cluster sampling, biodiversity conservation, endangered species, karst, red list, regeneration, tree ecology.

INTRODUCTION

Most tree species in tropical forests occur in relatively low abundance but account for the high levels of species richness (Condit, 1995). Rare species are often characterized by a restricted geographical range, high habitat specificity and small population size and/or abundance (Rabinowitz et al., 1986).

Thus, as a result of land use changes and over exploitation, some rare tropical tree species have become endangered due to their shrinking habitat and population size (Gaston, 1994; Nghia, 1999), which in many cases has led to species being red-listed (IUCN, 2010). In response, efforts have been made to acquire a better ecological understanding on rare threatened and endemic tree species' numbers, ecological niches and spatial distributions (Varghese and Menon, 1999) to inform appro-priate strategies for conservation.

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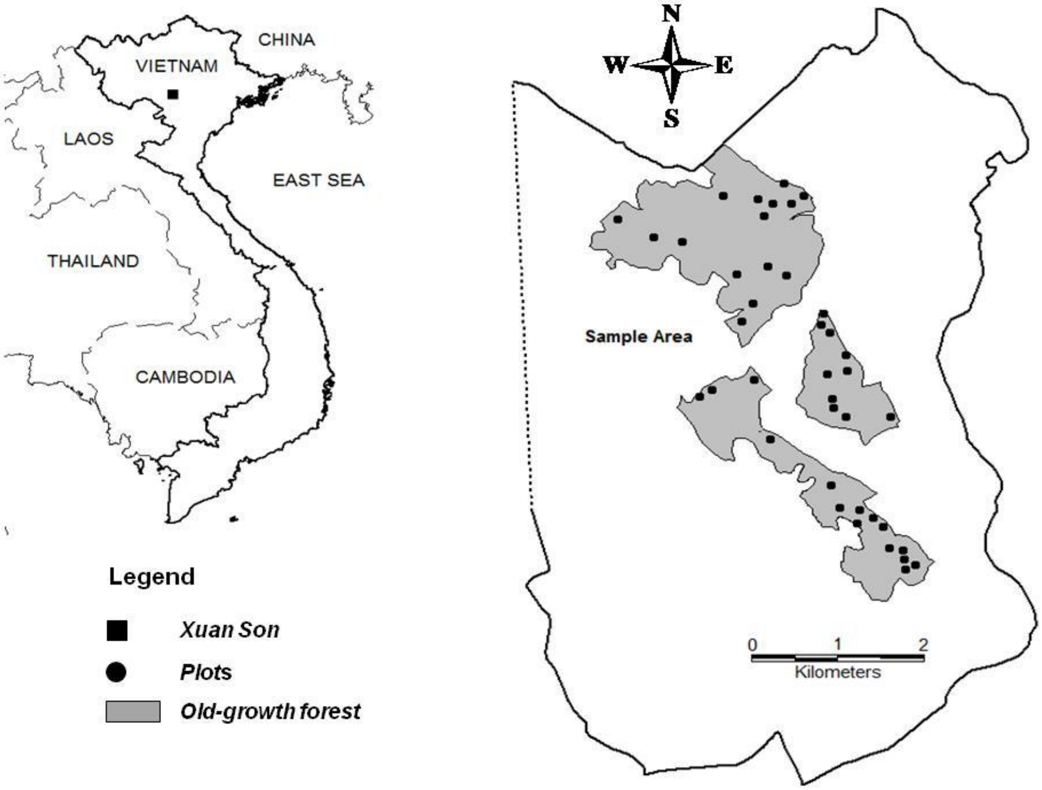

Figure 1. Location of the study area in the Xuan Son National Park and sample plots, northern Vietnam.

The flora of Vietnam has been estimated to comprise some 12,000 vascular plant species, of which ca. 30% are endemic. In northern Vietnam and southern China, the estimated proportion of endemics is even higher at 50% (Trung, 1998; Regalado et al., 2005). Limestone karsts contribute considerably to such high levels of plant diversity and forests thereon often host a high abundance of endemic species (Clements et al., 2006). In Vietnam, forests on Limestone Mountain mainly occur in the north (Phon et al., 1999). In limestone forests of the neighboring south-west China 1,394 vascular plant species (Zhu et al., 2003) and a total of 100 tree species (Tang et al., 2011) are recorded.

Limestone mountain forests can also be subject to transformation and/or exploitation. In response, some have been declared National Parks, such as the Xuan Son National Park in northern Vietnam. However, despite such high conservation status, little information exists on the abundance of rare tree species in such forests. In the present study, we consider five tree species listed in the Red List of Vietnam (Ban, 2007) occurring within the Xuan Son National Park. Three of the species (*Excentrodendron tonkinense* (Gagnep.) Chang & M, *Chukrasia tabularis* A. Juss. and *Garcinia fagraeoides* A. Chev.) are largely restricted to limestone mountain forests and considered typical to such environments, leading to them being regarded as specialists. The two other spe-cies (*Parashorea chinensis* H. Wang and *Melientha suavis* Pierre) have a wider ecological distri-bution range. The objectives of this study were: (i) to assess the abundance of these tree species and (ii) to provide information on potentially relevant ecological factors. The five selected tree species have a high market value and hence face overexploitation. As such, it is anticipated that our results may contribute toward a base-line against which the potential impact of logging may be assessed and appropriate management measures put in place as deemed appropriate.

MATERIALS AND METHODS

Study area

The study was conducted in the Xuan Son National Park (21° 03'-21° 12 N, 104° 51'-105° 01' E), which is located in Phu Tho province, northern Vietnam (Figure 1). The forested area of Xuan Son was established as a nature reserve in 1986 and declared as National Park in 2002. The Park covers an area of 15,048 ha and is surrounded by a buffer zone of 18,639 ha. The topography is characterized by many steep mountains, with altitudes ranging from 200 m to 1,386 m a.s.l. The area experiences a tropical monsoon climate with relatively cold winters spanning November to March and relatively hot summers; the mean annual temperature is 22.5°C and the annual average rainfall is 1,826 mm (Phu Tho, 2002). Approximately 17% of the National Park is comprised of lowland and lower montane evergreen forest on limestone, which are dominated by broadleaf evergreen tree species from the families Sapindaceae (*Amesiodendron chinense* (Merr.) Hu), Meliaceae

Table 1. Characteristics of the study species and their Red List categories.

Specie	Family	Main uses	Dispersal modes	Main threats	Vietnam Red List category (Ban, 2007)	IUCN Red List category (IUCN, 2010)
Excentrodendron tonkinense (Gagnep.) Chang & M	Tiliaceae	Timber, tannin extracted from bark	Ballistic[a]	High demand fortimber; overexploitation; habitat loss and fragmentation	EN	EN
*Garciniafagraeoides*A. Chev.	Clusiaceae	Timber	Gravity[b]	Scarce natural regeneration; overexploitation; habitat loss and fragmentation	EN	nl
Chukrasia tabularis A. Juss.	Meliaceae	Timber	Wind[c]	Overexploitation; habitat loss and fragmentation	VU	LR
Parashorea chinensis H. Wang	Dipterocarpaceae	Timber	Wind, Gravity[d]	High demand for timber; overexploitation; habitat loss and fragmentation	VU	EN
Melientha suavis Pierre	Opiliaceae	Vegetable, medicinal	Gravity[b]	High demand for vegetable; overexploitation; habitat loss and fragmentation	VU	nl

Status of threat indicated as follows: EN, endangered; VU, vulnerable; LR, lower risk; nl, not listed.
Sources: a, Wang (1986); b, not available and based on personal impression; c, Chat (1996); d, Lan et al. (2012).

(*Aphanamixis grandiflora* Blume) and Euphorbiaceae (*Cleidion spp.*) (Phu Tho, 2002; Hoi and Dang, 2008).

Site selection and study species

Field work was carried out in 2010 and 2011 in an old-growth forest area of 651 hectares at an altitude ranging from 300 m to 700 m a.s.l. The identification and delimittation of the study site was an essential initial step for randomly selecting the study plots and was based on a map of the vegetation status (FIPI, 2008) along with a reconnaissance survey. The studied old-growth forest is located within the strict protection zone of the National Park, which is exposed to little or no human disturbance. It is characterized by an abundance of trees of comparatively large diameter with no stumps being evident, despite some surrounding areas being formerly affected by logging.

The study focused on five rare tree species (Table 1) with the selection criteria requiring that they be of high conservation concern (that is listed in Vietnam's Red List) and easily identifiable in the field. Their occurrence was reported previously at the Xuan Son National Park (Hoi and Dang, 2008) and more detailed descriptions of the characteristics and red list categories of the species can be found in FIPI (1996), Chan and Huyen (2000), Ban (2007) and IUCN (2010).

Sampling strategy

Many tree species are not only rare but can also occur in spatially aggregated populations (Roesch, 1993; Acharya et al., 2000). Such aggregated distribution patterns dominate for tree species in karst forests, which are characterized by high habitat heterogeneity (Zhang et al., 2012). Hence, assessing their occurrence and spatial distribution by conventional sampling designs presents the problem of having many plots showing an absence of the sample trees, which most likely leads to an inappropriate evaluation of exactly how threatened any particular species may be. Adaptive cluster sampling (ACS) can however be employed as an appropriate method to overcome such shortcomings (Thompson, 1990; Thompson and Seber, 1996) and this method was consequently adopted for the current study.

First of all, the study area was overlain with a cell grid (30 x 30 m) overlain with a cell grid (30 x 30 m) delineated from a vegetation map for the Xuan Son. A total of 40 initial sample units were randomly selected from this grid for field investigations. Each initial unit consisted of an inventory plot of 30 x 30 m (900 m^2) for trees with diameter at breast height (DBH) ≥ 5 cm (hereafter referred to as tree$_{\geq 5}$) in addition to a smaller plot (10 x 10 m) located at the centre of a 900 m^2 plot for sampling regeneration (DBH < 5 cm and height ≥ 20 cm). The slope positions of these 40 randomly selected plots were then classified according to their relative location (lower, mid and upper slope) to better understand the distribution of the species.

In the 40 initial units, DBH, height, density of target species and other tree species were recorded. All non-target tree species were, as far as possible, identified with the assistance of the National Park parataxonomist and the Xuan Son flora list (Hoi and Dang, 2008). Individuals that could not be determined to species level were classified by their genera or families and sorted into specimens per plot to estimate the species richness per plot.

Rock outcrop exposure per plot (%) was visually estimated as a mean value for rock exposure across the 16 systematic subplots (each 5 x 5 m) taken as a proportion of the subplot area. Soil depth of each plot (cm) was determined as the mean of 25 systematic measurements taken using a metal stake set in the ground at five meter intervals. Slope (degree), aspect (degree) and altitude (m) were recorded for each plot and five hemispherical photographs were taken at regularly-spaced points within each plot using a digital camera (Minolta Dimage Xt, 185° fish-eye lens). Images were analyzed with the CanEye 5.0 software for canopy cover. These variables were selected because they are relatively simple to measure and reflect the features of the karst forest.

Based on the 40 initial plots, the ACS design was then applied to find more tree individuals and any regeneration of the five selected tree species and for each species separately. For each initial plot, where at least one individual of the target tree$_{\geq 5}$ was found, four neighboring plots of 900 m^2 were attached to the initial plot. Next, if any of these added plots contained at least one individual of the target species, further plots were added. The procedure was stopped only when no further plots hosted the specific target species. Thus, under the ACS design, the final number of sampled plots was not pre-determined but depended on the number of trees observed. For sampling regeneration, the same procedure was followed, but 100 m^2 neighboring plots were situated 30 m from the centre of the initial plot (Figure 1 in Appendix).

Two previously selected initial plots had to be replaced due to inaccessibility and safety concerns and the adaptive sampling was stopped at such sites. These inaccessible areas were, in most cases, rocky surfaces sparsely covered by herbs, shrubs and small trees in crevices.

On adaptively added plots, data was collected for the target species only and included the number of individuals as well as additional plot characteristics, including rock outcrop exposure and slope.

Statistical analysis

Detrended Correspondence Analysis (DCA) was used to identify ecological factors most likely associated with the densities of study species across the 40 initial plots with the software PC-ORD 5. Ten ecological variables (forest density, basal area, canopy cover, Shannon index, tree species richness, altitude, aspect, rock cover, soil depth and slope) were selected. However, the variables of soil depth and slope were removed due to the high correlation with rock cover. Spearman correlation was applied to check for correlations between density or ecological variables and axes scores with the significant level at $p \leq 0.05$. The Mann-Whitney test was used to test differences in median values of species density among three topographic positions. The non-parametric test was applied because the data included many zeroes and were not normally distributed. For ACS, we used the modified unbiased Hansen-Hurwitz estimator (Thompson and Seber, 1996) to calculate the mean species densities and their standard errors; inaccessible plots were treated as missing values.

RESULTS

Across the 40 initial sampling plots, *Cleidion javanicum* (Euphorbiaceae) was the most dominant and frequent tree species, followed by *Amesiodendron chinense* (Sapindaceae), *Aphanamixis grandiflora* (Meliaceae), *Terminalia myriocarpa* (Combretaceae), *Diospyros* spp. (Ebenaceae) and *Syzygium* spp. (Myrtaceae) respectively. Stand structural characteristics of the study forest are provided in Table 2.

On 13% of these plots, tree$_{\geq 5}$ individuals of *P. chinensis* and *M. suavis* were found, whereas *C. tabularis* and *G. fagraeoides* were much more frequent with 35 and 60%, respectively (Table 3). Densities of the five study species were relatively low and ranged from 0.4 to 3.7% for tree$_{\geq 5}$ individuals (Table 3).

For tree$_{\geq 5}$ individuals, the DCA axis-1 (eigenvalue = 0.76) showed significant correlations (Spearman at $p < 0.01$) with rock cover ($r = - 0.70$), canopy cover ($r = 0.69$), altitude ($r = 0.64$) and tree species richness ($r = 0.53$) (Figure 2). Axis 2 provided little information and was thus not included in the results.

With axis-1, densities of *E. tonkinense* and *C. tabularis* showed negative correlations ($r = - 0.68$ and $r = - 0.69$, respectively), and the density of *P. chinensis* showed a positive correlation ($r = 0.71$) (Spearman at $p < 0.01$). We also found similar impacts of the above-mentioned variables on densities of regeneration, with the strongest influence being on the variable tree species richness.

Rock-outcrop cover and steepness increased from the lower to upper slope positions, (median, from 58 to 93% and from 31 to 43 degrees, respectively) while soil depth decreased (median, from 13 to 3 cm) (Table 1 in Appen-dix). Hence, densities of tree$_{\geq 5}$ individuals of *E. tonkinense* and *C. tabularis* were significantly higher on upper slope than on mid-slope positions, while they were not found on the lower slope positions. The density of *G. fagraeoides* did not differ significantly among the three topographic positions, but seemed to be very heterogeneous. In con-trast, densities of *P. chinensis* and *M. suavis* were higher on lower slope than on mid-slope positions, and there was no evidence for the two species on the upper slope (Table 4). Similar results were also found for regeneration of the five study species (Table 2 in Appendix).

In the networks of ACS, numbers of individuals of all five species recorded were considerably higher than expected for both tree$_{\geq 5}$ individuals and regeneration (Table 5). *G. fagraeoides* density was highest, while lower values were recorded for *E. tonkinense* and *C. tabularis* and the lowest for *P. chinensis* and *M. suavis*.

We found that the maximum density of tree$_{\geq 5}$ individuals of *E. tonkinense*, *C. tabularis* and *G. fagraeoides*, increased with increasing rock outcrop cover (Figures 3a to c). However, within a given class of rock outcrop cover, there was very high variability in tree density. *E. tonkinense* and *C. tabularis* only occurred on sites with rock outcrop exposure above 75%, while *G. fagraeoides* occurred on sites with over 45%. The tree$_{\geq 5}$ individuals of *P. chinensis* indicated a decreasing trend of density with increasing rock outcrop exposure (Figure 3d). Both *P. chinensis* and *M. suavis* did not occur on mostly rocky sites (Figures 3d and e).

The regeneration of two out of three typical limestone species (*C. tabularis* and *G. fagraeoides*) increased in maximum density with higher rock outcrop exposure

Table 2. Stand structural characteristics of old-growth forest across the 40 initial plots.

Parameter	Median	Minimum	Maximum
Stem density (≥ 5 cm, trees/ha)	573	389	811
Diameter (cm)	17.6	10.7	23.5
Height (m)	9	5.9	12.6
Basal area (m^2/ha)	18.9	6.8	33.2
Canopy cover (%)	74	64	83
Tree species richness (no. per 900 m^2)	24	19	32
Exposure of rock outcrops (%)	88	45	95
Soil-depth (cm)	5.5	0.8	19.6
Slope (degree)	40	20	49
Altitude (m a.s.l.)	487	318	680

Table 3. Frequencies and stem densities of the five study species across the 40 initial plots.

Specie	Tree≥5		Regeneration	
	Frequency (%)	Stem density (%)	Frequency (%)	Stem density (%)
E. tonkinense	28	1.3	20	0.7
C. tabularis	35	1.8	28	1.0
G. fagraeoides	60	3.7	50	2.1
P. chinensis	13	0.5	18	0.5
M. suavis	13	0.4	13	0.4

(Figures 3g and h). *P. chinensis* regeneration showed an opposite trend, and both *P. chinensis* and *M. suavis* did not occur on mostly rocky sites (Figures 3i and j). *E. tonkinense* regeneration only occurred on very rocky sites (Figure 3f).

DISCUSSION

The five study species are categorized as threatened tree species in the Red List of Vietnam (Ban, 2007). However, our study shows a high abundance of mature and regenerating *G. fagraeoides* trees and also a considerable number of *C. tabularis* and *E. tonkinense* trees across the study region. *P. chinensis* and *M. suavis*, the species not strictly linked to limestone mountain forests, had low densities. A considerable number of *G. fagraeoides*, *C. tabularis*, *P. chinensis* and *M. suavis* individuals were also found in the Ben En National Park in north-central Vietnam (Hoang et al., 2008; Hoang et al., 2011); however, they were classified as 'threatened' due to the danger of illegal logging (Hoang et al., 2008). While *E. tonkinense* was not found in the Ben En

National Park, in south-west China, the *E. tonkinense* population was recorded as having significantly declined as a result of human intervention and its natural regeneration is no longer supported in some disturbed habitats (Tang et al., 2005).

Our results are therefore inconsistent with some published information on endangerment levels of these species in Vietnam (FIPI, 1996; Chan and Huyen, 2000; Ban, 2007), notwithstanding the fact that a species may be rare at a given spatial scale but not at another (Gaston, 1994). Assuming rarity of our study species at a larger scale, for example across northern Vietnam, we conclude that the National Park under study and its core area are well placed in respect of facilitating the conservation of the five study species. However, it cannot be completely ruled out that more than the expected number of individuals of these species may also exist in other areas without having been formerly detected, or as a result of less effective sampling procedures.

Vegetation patterns, including tree species composition, commonly vary depending on topographic position from lower to upper slope positions (Kikuchi and Miura, 1993; Aukema et al., 2007). In this study, *E. tonkinense* and *C. tabularis* were mostly found on upper and mid slopes of hills, and they hardly occurred in lower slope positions, valleys or foothills across the study region. *E. tonkinense* can grow on bare rock, steep slopes, in shallow soil and on extremely dry sites (Wang et al., 1986; Tang et al., 2005). *C. tabularis* can cope with a wider range of soil conditions, but it is most commonly observed on limestone, which is generally more light-textured and well-drained (Chat, 1996). Densities of *E. tonkinense* and *C. tabularis* had significant negative

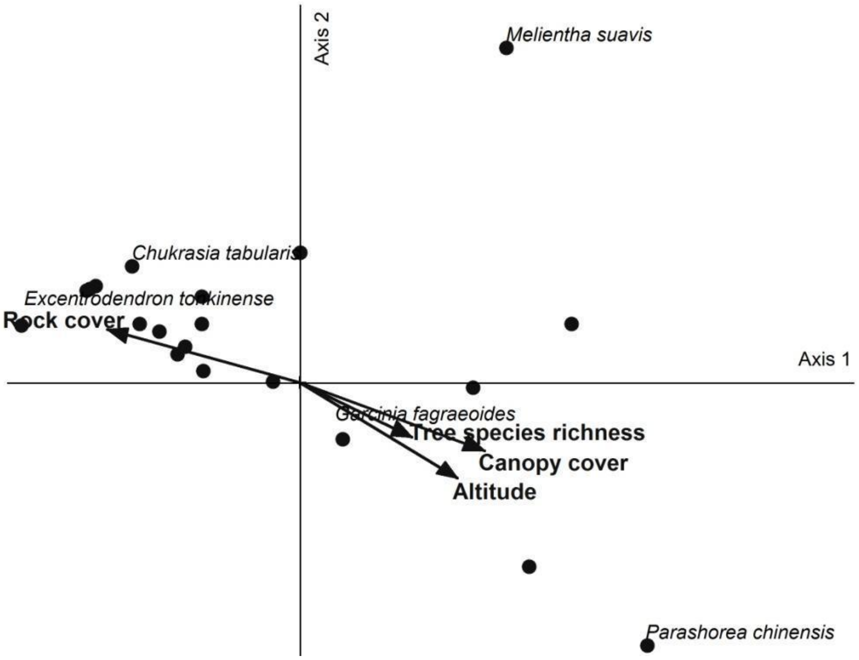

Figure 2. DCA diagram showing the relationship between tree$_{\geq 5}$ densities and the ecological variables. The matrix consisted of 33 initial plots (7 of 40 initial plots showed no occurrence for any of the target individuals and were therefore excluded from the matrix), 5 tree species, rock cover, canopy cover, altitude, and tree species richness. Axis 1: eigenvalue = 0.76, explained variance = 60%; correlation threshold $r^2 > 0.31$; vector scaling 126.

Table 4. Stem density (no. per ha) of tree$_{\geq 5}$ individuals of the five study species on three topographic positions in the random sampling design. Differing small letters indicate statistical differences among slope positions in the Mann-Whitney test ($p \leq 0.05$).

Specie	Lower slope (n = 6)			Mid-slope (n = 20)			Upper slope (n = 14)		
	Median	Minimum	Maximum	Median	Minimum	Maximum	Median	Minimum	Maximum
E. tonkinense	0^a	0	0	0^a	0	44	11^b	0	67
C. tabularis	0^a	0	0	0^a	0	44	22^b	0	56
G. fagraeoides	0^a	0	22	22^a	0	78	11^a	0	78
P. chinensis	17^a	0	33	0^b	0	0	0^b	0	0
M. suavis	0^a	0	44	0^{ab}	0	22	0^b	0	0

correlations with species diversity and canopy cover of forests. This fact may be explained by two mechanisms of traditional niche-assembly theories, namely, environmental constraints and competitive displacement (after Chave, 2008). *E. tonkinense* and *C. tabularis* are capable of growing on severe sites, albeit relatively slowly, and on better sites, they are most likely out-competed by faster-growing species. These patterns would then be somewhat similar to the case of the limestone species *Kunzea sinclairii* in New Zealand, which can expand its distribution range but is often excluded in the regeneration process by interspecific competition (de Lange and Norton, 2004). *G. fagraeoides* was the most abundant of the five studied tree species, with increasing occurrence from lower to upper slopes, where its growth on rocky sites is supported by strongly - developed roots that hold onto rock and penetrate deeply into rock fissures (FIPI, 1996).

In contrast to the three specialist tree species, *P. chinensis* was less abundant on upper slopes and preferred lower slopes or valleys. Its distribution was probably spatially aggregated (Tang, 2007) because its seeds are relatively weighty, resulting in a dispersal range near the conspecific adults (Lan et al., 2012). Moreover, a long breeding period, recalcitrant seeds and very low seedling survival probably make it difficult for the *P. chinensis* population to spread (Shouhua and Jiangua, 1990; Nghia, 1999).

We found an increasing number of *P. chinensis* trees with increasing tree species diversity and canopy cover, which may point to positive interactions, as suggested by

Table 5. Total number of plots assessed, total number of individuals detected and stem density (Hansen-Hurwitz estimator) of the five study species in adaptive cluster sampling.

Species	Tree$_{\geq 5}$			Regeneration		
	Total number of plots (n)	**Total number of individuals (n)**	**Stem density Mean ± SE (number per ha)**	**Total number of plots (n)**	**Total number of Individuals (n)**	**Stem density Mean ± SE (number per ha)**
E. tonkinense	147	210	11 ± 3	106	73	45 ± 15
C. tabularis	188	237	12 ± 3	132	95	66 ± 20
G. fagraeoides	378	516	22 ± 4	244	176	110 ± 19
P. chinensis	110	59	3 ± 2	85	34	29 ± 13
M. suavis	95	47	3 ± 1	68	20	24 ± 12

Lan et al. (2012). Sites with deep soils and high soil moisture seem to be favorable habitats for the dipterocarp species (FIPI, 1996; Nghia 1999). Lan et al. (2012) also found a negative association between *P. chinensis* density and gaps in the tro-pical forest of Xishuangbanna in south-west China.

M. suavis was generally found in the mid or lower sections of slopes but rarely on the upper sections, which is consistent with that found for *M. suavis* populations in northern Thailand (Khamyong and Seramethakun, 1996). The authors concluded that the very short viability of seeds combined with dry site conditions results in poor natural regene-ration. Our results lend support to this conclusion in that we mainly found *M. suavis* regeneration under areas of high canopy cover and on diverse plots.

Our use of ACS proved to be efficient as we found more trees per unit of time using the method (on average 27 minutes per individual) than was the case with random sampling (on average 44 minutes per individual). This was also influenced by the steep and rocky study area, which made movement between distant random plots difficult.

Based on our current knowledge, we recommend including a clear 'stop rule' (restricted ACS) as some species, such as *G. fagraeoides,* were more frequent than expected, which lead to possible over-sampling. ACS also restricts statistical analysis in that sampled plots are not independent from one and other, where a higher number of tree species is being studied, it will also increase the time needed in comparison with random sampling. Nonetheless, by using ACS, we encountered more individuals and generated additional information on sites where the five study species actually occurred.

For our study, ACS sampling revealed a high variation in tree densities of the specialist species (*E. tonkinense, C. tabularis* and *G. fagraeoides*) among plots (Figure 3), which remains unexplained by the ecological factors assessed. Based on our observations and those by Doi (2002), this variation might partly be caused by the ruggedness of the rock surface and the size and shape of the stones, which also influence the quantity of finer soils. Our study species were often found on rocks with a rugged surface, where seeds may be assumed to have germinated within rock fissures or holes with saplings and trees being able to develop robust roots that provide stability and penetrate deeply into fissures (Figure 4). However, where the rock-outcrop surface was completely bare or if little soil was available in the surrounding area, no trees were growing, which is indicated in the occasional zero density record for specialist species in Figure 3.

Conclusion

The studied limestone forest specialist species(*E. tonkinense*, *C. tabularis* and *G. fagraeoides*) are most likely rare at larger scales, such as across northern Vietnam and beyond; however, they were found with considerable abundance in the studied old-growth forest of the Xuan Son National Park. The species show a strong site preference towards steep slopes, shallow soil and high rock outcrop cover, a conclusion which is also supported by the relative abundance of regeneration. These specialist species may therefore play an important role in the stabilization of steep slopes or in the restoration of sites where little fine earth is found, thus highlighting a potential use outside protected areas. The two other study species (*P. chinensis* and *M. suavis*), which are known to have wider distribution ranges; do occur in this habitat, but not on mostly rocky sites.

Based on the relatively high abundance of the red list species recorded at the study site, we consider

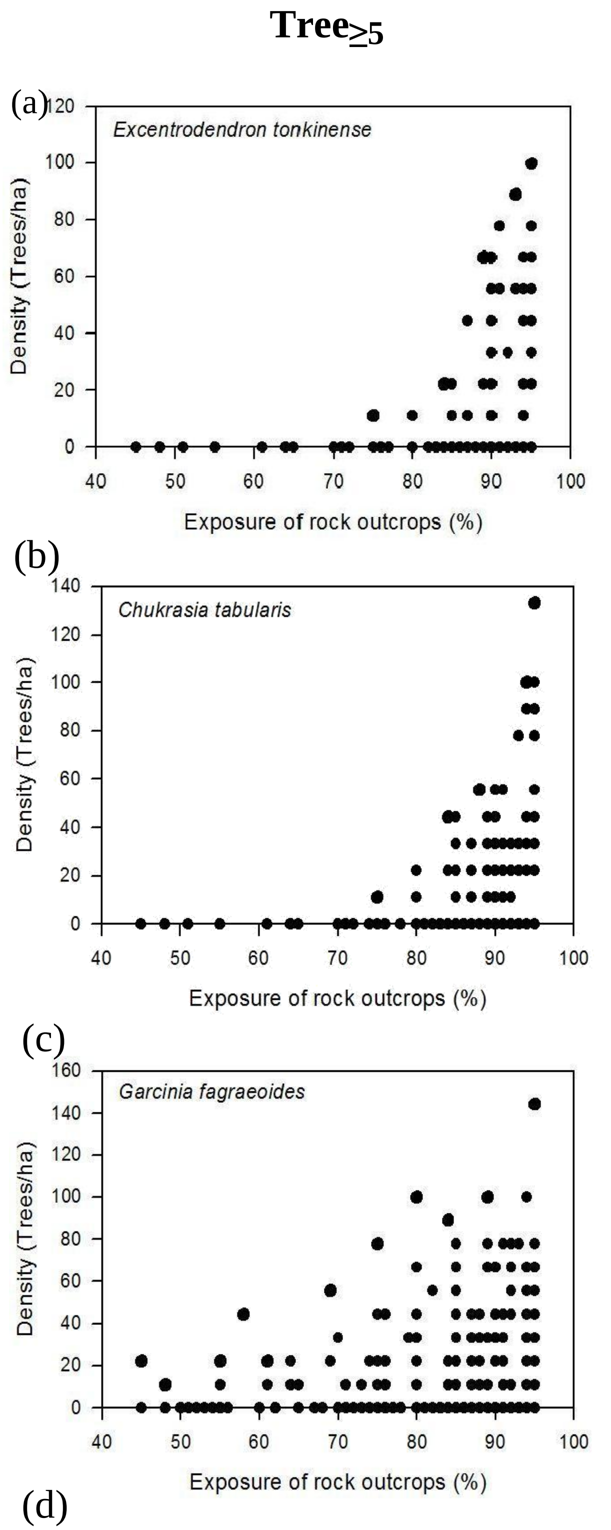

Figure 3. Relationships between species densities and rock outcrop exposure for five study species. From (a) to (e) for trees with DBH ≥ 5 cm (tree≥5) and (f) to (j) for tree regeneration.

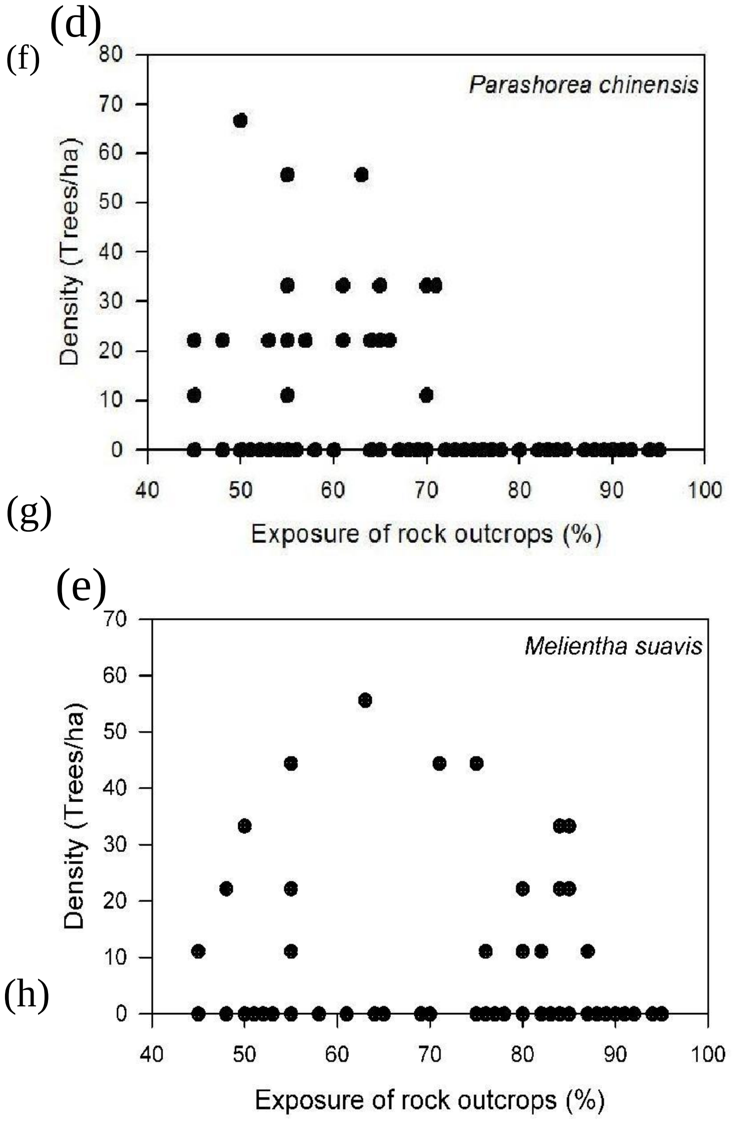

Figure 3. Contd.

it to be an important refuge, particularly in view of former and continuing over exploitation and degradation of such habitats and species of importance. Our results may there-fore contribute toward any baseline assessment aimed at addressing impacts of logging and management. The high and partly unexplained heterogeneity of densities of some species will however make it difficult to delineate management effects.

(i)

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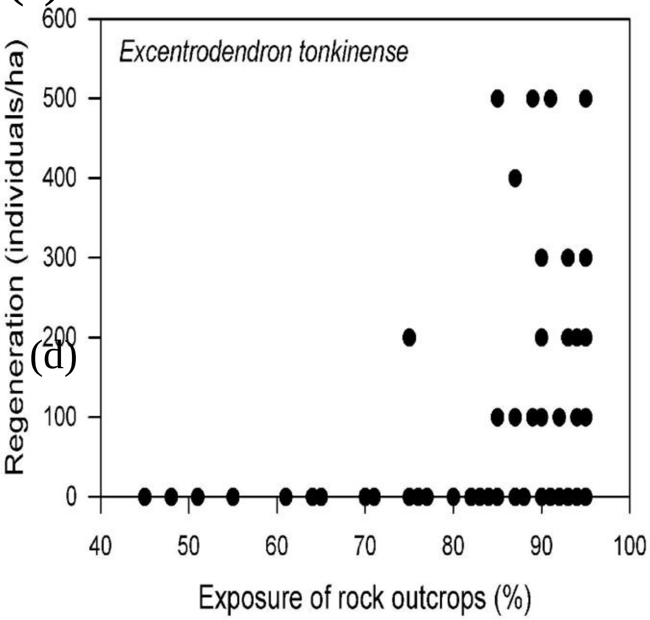
(c)

Regeneration

(f)



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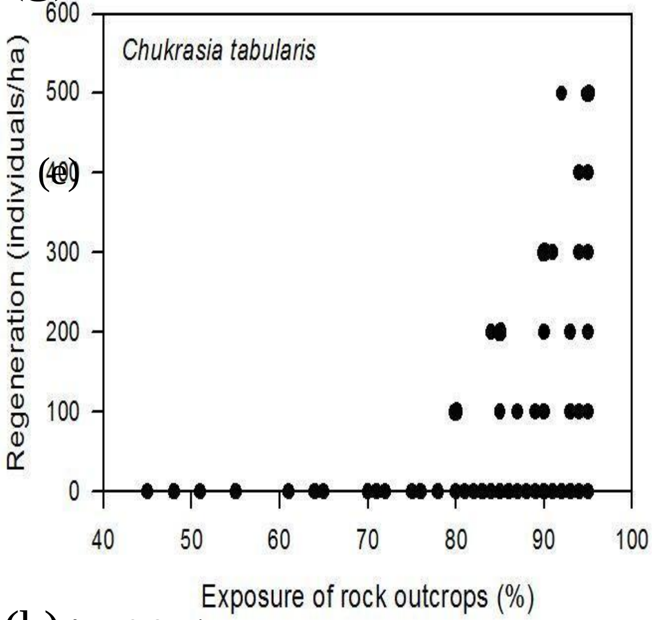

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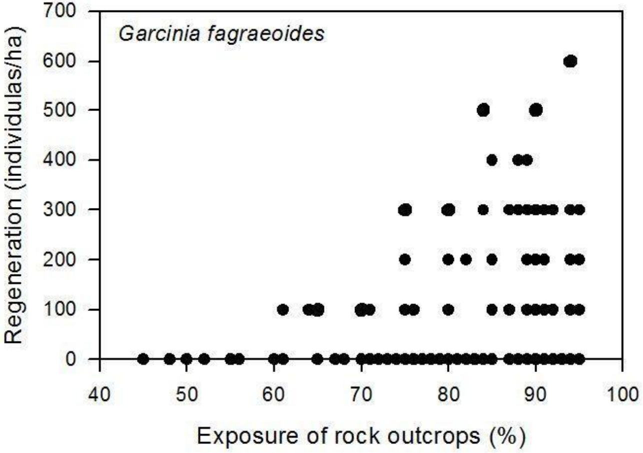

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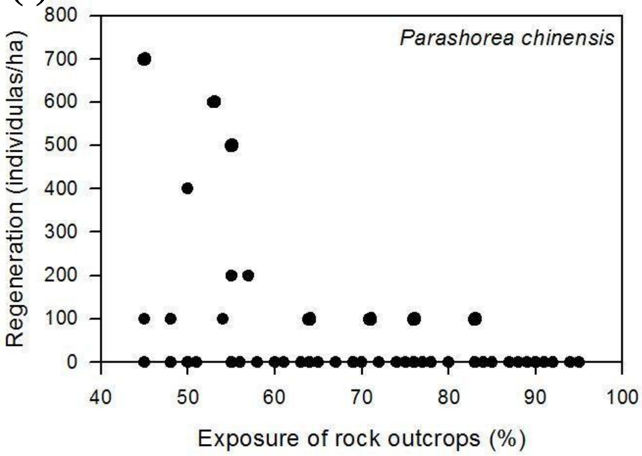

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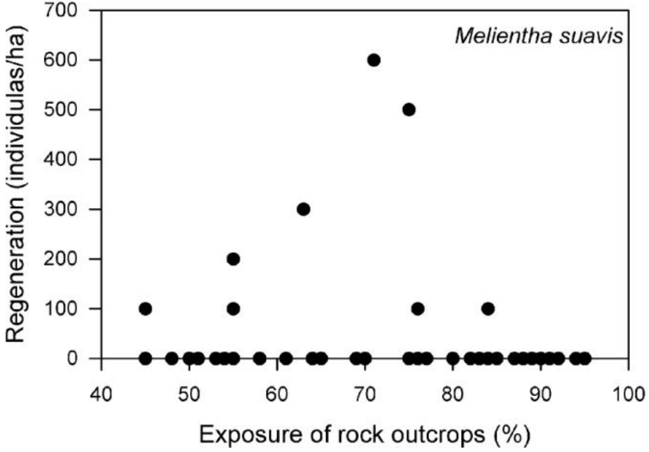

Figure 3. Contd.

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Figure 4. *G. fagraeoides* on the rocks: a small and a bigger statured tree (left) and a seed (right).

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APPENDICES

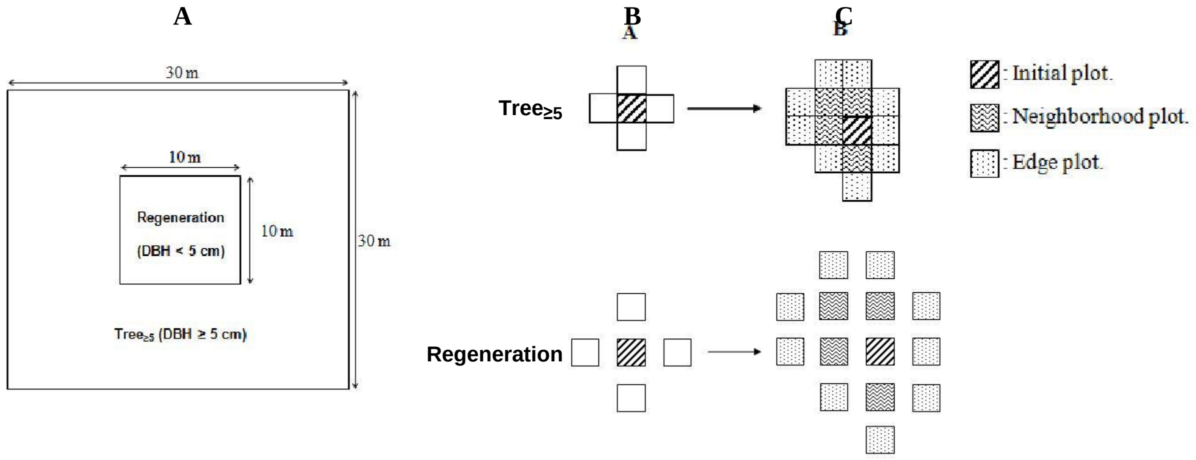

Figure 1. Plot design of the adaptive cluster sampling. (A) The initial unit consisted of an inventory plot for tree$_{\geq 5}$ and of a smaller plot for regeneration. (B) The procedure of adaptively adding neighborhood plots for tree$_{\geq 5}$ and regeneration. (C) The cessation of adaptive sampling when a cluster was surrounded by edge plots without target individuals.

Table 1. Comparison of rock cover, soil depth and slope among three topographic positions. Differing small letters indicate statistical differences among slope positions in the Mann-Whitney test ($p \leq 0.05$).

Parameter	Lower slope (n = 6)			Mid-slope (n = 20)			Upper slope (n = 14)		
	Median	Minimum	Maximum	Median	Minimum	Maximum	Median	Minimum	Maximum
Rock cover (%)	58^a	45	71	87^b	51	92	93^c	80	95
Soil depth (cm)	13^a	11	20	6^b	3	15	3^c	1	8
Slope (degree)	31^a	20	35	40^b	28	49	43^c	34	49

Table 2. Stem density (no. per ha) of regeneration of the five study species on three topographic positions in the random sampling design. Differing small letters indicate statistical differences among slope positions in the Mann-Whitney test ($p \leq 0.05$).

Specie	**Lower slope (n = 6)**			**Mid-slope (n = 20)**			**Upper slope (n = 14)**		
	Median	**Minimum**	**Maximum**	**Median**	**Minimum**	**Maximum**	**Median**	**Minimum**	**Maximum**
E. tonkinense	0^{a}	0	0	0^{ab}	0	400	0^{b}	0	300
C. tabularis	0^{a}	0	0	0^{a}	0	200	50^{b}	0	500
G. fagraeoides	0^{a}	0	100	50^{ab}	0	500	100^{b}	0	600
P. chinensis	100^{a}	0	500	0^{b}	0	100	0^{b}	0	0
M. suavis	50^{a}	0	600	0^{b}	0	100	0^{b}	0	0