

Vegetation Structure and Floristic Composition along the Edge of Montane Forest and Agricultural land in Um Phang Wildlife Sanctuary, Western Thailand

Dokrak Marod^{1, 2,*}, Lamthai A-sanok³, Prateep Duengkae¹ and Anak Pattanavibool⁴

ABSTRACT

The impacts of agricultural practices on vegetation changes along forest edges were studied by the selection of appropriate native species for reforestation in degraded lower montane forest (LMF) in the Um Phang Wildlife Sanctuary, Western Thailand. Three different land use practices, abandoned area (AB), annual cropping (AC) and shifting cultivation (SC), were selected for studying the changes in vegetation structure and species composition along a gradient from the interior of LMF into adjacent agricultural lands. At each land use site, three belt transects of 20 × 150 m were established perpendicular to the forest-agriculture edge. The vegetation transect census recorded the living numbers of seedlings, saplings and trees, excluding lianas, and the stem diameter and height were measured for saplings and trees. The light and soil conditions in every belt transect were also investigated.

In total, 78 tree species in 73 genera and 40 families were found across all sites. The Shannon-Weiner diversity index (H') values were significantly higher in the remnant forest (RF) than near-to-edge (nAL) and distant-from-edge (dAL) and the species composition at the forest edge consisted mainly of secondary shrubs and tree species. From the RF toward the edge, numbers of seedlings, saplings and secondary trees increased, while numbers of mature forest trees declined. Environmental factors—namely, relative light intensity (RLI), soil moisture content (SMC) and soil bulk density (SDb)—were significantly ($P < 0.05$) different along the forest edge. The plant species, *Shima wallichii*, *Eurya accuminata*, and *Collona elobata*, had a high correlation that was intermediate between the RLI and SMC gradients, especially in nAL. Thus, these species should be strongly supported in programs associated with the ecological restoration of LMF.

Keywords: Tropical montane forest, edge effect, fragmentation, regeneration, ecological restoration

INTRODUCTION

Forest fragmentation and deforestation have become critical environmental problems with possible implications on a global scale (Turner

and Meyer, 1991; William and Richard, 1997), as anthropogenic activities convert continuous forests into landscapes of remnant forest (RF) patches, embedded in agriculture matrices (Green and Sussman, 1990; Dunn *et al.*, 1993). In tropical

¹ Department of Forest Biology, Faculty of Forestry, Kasetsart University, Bangkok 10900, Thailand.

² Center for Advanced Studies in Tropical Natural Resources, Kasetsart University, Bangkok 10900, Thailand.

³ Department of Agroforestry, Maejo University Phrae Campus, Phrae 54140, Thailand.

⁴ Wildlife Conservation Society Thailand Program, Nonthaburi 11120, Thailand.

* Corresponding author, e-mail: dokrak.m@ku.ac.th

forests, vast expanses of original forest have been converted into pasture and agricultural fields, often under shifting cultivation (SC) with subsequent abandonment of the temporary sites (Brown and Lugo, 1990; Chapman and Chapman, 1999). The effect of the conversion is the formation of edges and the modification of pre-existing edges between vegetation types, an increase in forest edges, and the subdivision of large forest areas into smaller non-contiguous fragments (Laurance, 2000). In addition, forest edge effects have been recognized as one of the most pervasive and deleterious processes occurring in the tropics (Murcia, 1995; Gascon *et al.*, 2000).

Forest edges can be considered a transitional ecosystem or buffer zone between the forest and the surrounding open areas (Kapos, 1989; Young and Mitchell, 1994; Laurance *et al.*, 1997; Williams-Linera *et al.*, 1998; Didham and Lawton, 1999). In tropical highland ecosystems, different kinds of forest edges had been able to develop, influencing the vegetation structure and composition (Mucia, 1995; Kapos *et al.*, 1997; Benitez-Malvido and Martinez-Ramos, 2003; Barlow *et al.*, 2006; Cagnolo *et al.*, 2006). Microclimatic variables (light intensity and duration, relative humidity and air temperature) and soil factors (pH, organic carbon, total nitrogen, available phosphorus, soil moisture and temperature) differ along the forest edges (Lovejoy *et al.*, 1986; Williams-Linera *et al.*, 1998), increasing the rates of tree mortality (Laurance *et al.*, 2002; Nascimento and Laurance, 2004).

The relationships between species traits, landscape attributes and species distribution have been the topic of several studies (Henly *et al.*, 2004; Kolb and Diekmann, 2005; Lindborg, 2007; Julien *et al.*, 2011). Generally in tropical forests, light requirements and the tolerance of seedlings to shading are the major traits that have been used to classify forest tree species into two ecological groups: light demanding (or pioneer) species and shade-tolerant (or late-succession) species (Bazzaz

and Pickett, 1980; Swaine and Whitmore, 1988). Usually, pioneer plant species are more abundant along the forest edges compared to shade-tolerant species (Chazdon *et al.*, 1996; Marod *et al.*, 2004), due to germination stimulated by light exposure (Ranney *et al.*, 1981; Aide and Cavelier, 1994). Thus, identifying common life-history traits that affect the sensitivity of species to environmental change, especially anthropogenic disturbance (Whitmore, 1991; Catherine and Jennifer, 2009; Julien *et al.*, 2011), is very important for managers wanting to develop conservation programs.

Despite the over-exploitation and the large amount of forest edge formation, there have been relatively few studies on the regeneration pathways along edges, and especially, little is known about the link between resource use at the local community level and its effects on forest fragmentation and loss at the landscape scale (Turner and Meyer, 1991). Knowledge of these patterns may also be useful in answering questions related to the long-term sustainability of human-forest interactions and for developing management policies that protect and enhance tropical forests, especially programs aimed at forest restoration (Vieira and Scariot, 2006). Enhancing a general understanding of how edge dynamics influence forest regeneration and ecosystem services on the micro-scale in tropical regions is important. Interactions between environmental conditions on forest edges and plants may determine thresholds beyond which the buffering capacity of the system is no longer strong enough to support either the restorative functions of forest regeneration or the ecosystem.

The present study examined differences in the vegetation structure and species composition along a forest interior-edge-exterior gradient under three different agricultural practices, AB, SC and AC. All the studies were conducted in fragmented lower montane forest (LMF, 1,000–1,100 m above sea level; a.s.l.). The following were considered regarding agricultural practices

and plant dynamics: (1) the impact of land use practices on vegetation structure and species composition and (2) how the species traits changed along the forest edge.

Study area

The study was carried out in fragmented LMF in the Um Phang Wildlife Sanctuary, Tak province, western Thailand (Figure 1) located between 98°33' and 99°07' E and between 15°33' and 16°23' N. The topography is generally mountainous with a maximum altitude of 2,152 m a.s.l. The study site included a network of many permanent rivers and streams dividing the area into valleys and lowland plains. The climate is seasonal with three distinct seasons—a cool dry season from November to February, a hot dry season from March to May and a rainy season from May to October. The highest rainfall occurs in July and August and the average annual rainfall is 1,445.7 mm. The mean annual temperature ranges from 23.3 to 28.0 °C and December is the coolest month when overnight temperature can drop to 2 °C, while April is the hottest month, with a maximum temperature of 40 °C. The main prevailing vegetation types are hill evergreen forest or

montane forest, followed by dry evergreen forest, mixed deciduous forest, dry dipterocarp forest and other successional forest types originating from disturbed forest, such as bamboo forest, savanna forest and grassland (Martin, 2003).

Several villages were located along the mountainside in the eastern forested area. SC practiced by hill tribes was the main cause of fragmentation as hill tribe cultivation results in SC with a 4–5 yr interval. However, some large patches with relatively undisturbed LMF still remained. LMF in Thailand is characterized by the families Fagaceae, Myrtaceae, Lauraceae, Theaceae and Magnoliaceae (Martin, 2003).

MATERIALS AND METHODS

Site selection

The study was conducted during July 2005–June 2006 at three representative sites selected on the basis of the presence of LMF patches adjacent to three different agricultural land practices—AB, AC and SC—with time periods since abandonment of 12, 1 and 5 yr, respectively. The selected sample sites were randomly selected within the homogeneous environmental factors of

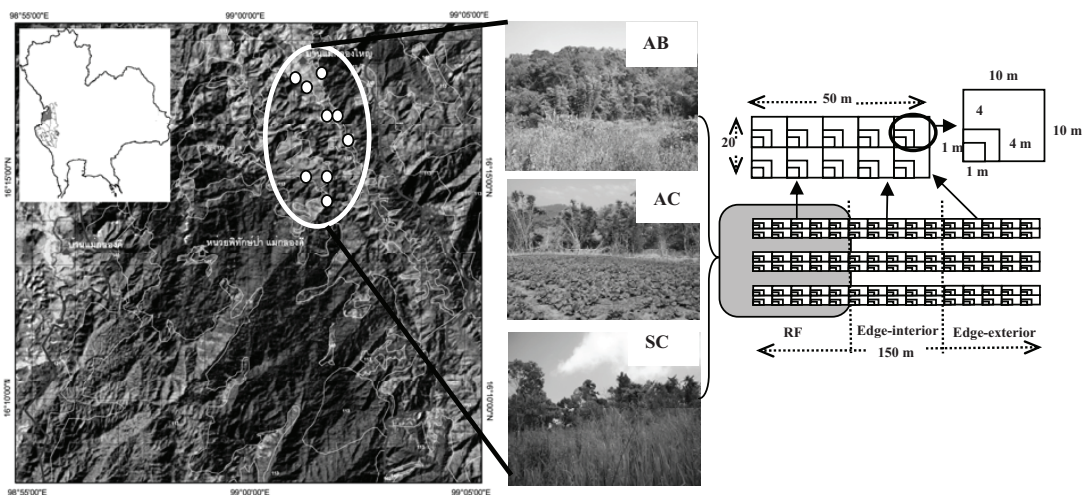


Figure 1 Three sampling belt-plots were established along the remnant forest (RF) forest edge of abandoned area (AB), annual cropping (AC) and shifting cultivation (SC) land uses.

altitude (1,000–1,100 m a.s.l.) and slope (55–65%). In addition, the connectivity among fragments was high due to the nearby presence of shredded, but contiguous relatively undisturbed LMF within a range of 600–800 m for the three sites.

Data collection

Following Bullock (1996), at each representative site, three sampling belt-plots sized 20×150 m were laid out (Figure 1), with a total of nine belt transects perpendicular to the forest edge laid out from the RF through the edge (0 m), to the exterior (that is, the agricultural land, from 0 to 100 m). Each transect consisted of 30 contiguous 10×10 m frame quadrats distributed over RF (10 quadrats) and agricultural land vegetation (20 quadrats), with a total of 90 frame quadrats on each site. In one corner of each 10×10 m frame quadrat, a subquadrat of 1×1 m was laid out within a single 4×4 m quadrat for woody stem analysis. Diameter at breast height (DBH) was defined as being 1.30 m above ground level. Living woody stems of trees and shrubs were counted, excluding lianas, identified and measured for three distinct stem-size categories in different-sized quadrats: ‘seedling’, (total height less than 50 cm) in the 1×1 m subquadrats; ‘sapling’, (DBH less than 5 cm, total height over 130 cm) in the 4×4 m subquadrat and ‘tree’ (DBH greater than or equal 5 cm, total height over 130 cm) in the 10×10 m quadrats. In addition, the number of herbaceous plants was recorded and identified in all 1×1 m subquadrats. DBH was measured for all saplings and trees. Total height was estimated for all recorded woody stems, using a clinometer. The lowest living branch and the average crown diameter were measured for all trees. The average crown diameter was estimated for each tree as the average of two perpendicular crown diameters projected on the forest floor.

Light conditions within the 10×10 m frame quadrat were estimated using hemispherical canopy photographs at 1 m height above ground with a fish-eye lens in December 2005 in the early

morning (before 1100 hours) in 270 photos. The relative light intensity (RLI, %) was analyzed by the FEW52b program (Ishizuka and Kanazawa, 1991). The RLI in each frame quadrat was estimated as the percentage of standard overcast sky distribution. Soil samples were collected from each subquadrat (10×10 m), producing a total of 270 samples. The soil bulk density (SDB, g.cm^{-3}) and soil moisture content (SMC, %) were analyzed from the soil samples which were collected from the topsoil layer (0–25 cm) from November to December 2005 during sunny weather.

Data analysis

Each set of 10 continuous 10×10 m quadrats was combined into one plot, resulting in nine 0.1 ha plots at each of the three representative sites—namely, remnant forest (RF, -50 to 0 m), near-to-edge agricultural land (nAL, 0 to +50 m), and distant-from-edge agricultural land (dAL, +50 to +100 m).

The dominant species in the area were evaluated using the importance value index (IVI) which was calculated as the sum of the relative density, relative dominance in basal area and relative frequency. A higher IVI value indicates greater dominance of that species on the site (Parrotta *et al.*, 1997). A vegetation data matrix of the ‘tree’ category (presence or absence using the IVI of each species), comprising nine belt transect plots (total 270 subquadrats) was constructed (Kent and Coker, 1992; Jongman *et al.*, 1995) to evaluate the species distribution in the multidimensional space represented by the various environmental parameters (RLI, SMC and SDB). Canonical correspondence analysis (CCA) ordination was conducted using the program PC-ORD, version 5.10 (McCune and Mefford, 2006) to determine the importance of the environmental variables along RF toward the exterior. In order to illustrate the spatio-temporal forest edge succession dynamics, a distribution map of some pioneer and shade-tolerant species was developed

encompassing a canopy species according to their relative abundance of stems (included 'tree' and 'sapling' categories).

In each of the 270 plots, the stem density (number of stems per area) was calculated for seedlings, saplings and trees. Forest-to-agricultural land vegetation profiles were drawn to illustrate changes in the vegetation structure from RF into nAL and dAL, respectively. Similarly, the basal area was calculated for stems with a DBH greater than or equal to 5 and 10 cm, respectively.

The plant species diversity was calculated at each site using the Shannon-Wiener diversity index, H' , (Magurran, 1988) as shown in Equation 1:

$$H' = -\sum_{i=1}^s p_i * \ln(p_i) \quad (1)$$

where p_i = the proportional cover of the i^{th} species ($i = 1, 2, 3, \dots, s$) in a plot.

Analysis of variance (ANOVA) was used to evaluate differences in the stem density, basal area, species diversity and environmental conditions among the three sites (Sokal and Rohlf, 1981).

RESULTS

Vegetation structure and species composition

In total, 78 tree species in 73 genera and 40 families were found across all three sites (Appendix A). The maximum canopy height inside RF was similar for all three sites, ranging from 17 to 20 m (Table 1). Plots were dominated by the genera *Castanopsis*, *Michelia*, *Betula* and *Shima*. On the other hand, the vegetation height in nAL and dAL of the AC and SC plots fluctuated widely (6–18 m), due to the occasional presence of a few scattered, isolated trees (such as *Michelia floribunda*, *Castanopsis acuminatissima* and *Quercus* spp.) in the low herbaceous grassland vegetation (Table 1 and Figure 2). A shrub layer of pioneer species such as *Rhus chinensis* and *Chromolaena odorata* was characterized in nAL and dAL, ranging in height from 1 to 3 m and being

taller in nAL than dAL for all sites (AC, SC and AB). Isolated remnant trees were found only at the SC and AB sites.

The stem density of trees, saplings and seedlings varied within the forest edge, RF, nAL and dAL among the land use practices. The tree density was significantly (ANOVA, $P < 0.05$) different in RF and nAL but not in dAL. In RF, the tree density was higher in SC than AB and AC, while in nAL the AB had a higher value than in either AC or SC (Table 1 and Figure 3). Sapling density was also significantly (ANOVA, $P < 0.05$) different in all three forest edge sites. In RF, sapling density in SC had a higher value than in AB and AC, while in contrast, in the exterior, nAL and dAL, lower values were found in AC than in AB and SC (Table 1 and Figure 3).

Seedling density showed significant (ANOVA, $P < 0.05$) differences among land use practices only inside RF and dAL. In RF, the seedling density was higher than in SC, AC and AB, respectively. In dAL, seedlings were detected in two land practice groups, AB-SC and AC, with the lowest number found in AC.

The vegetative regeneration along RF toward the exterior was also analyzed in each land use practice. In AB, trees and saplings were significantly (ANOVA, $P < 0.05$) different; however, there was no statistically significant effect detected for seedlings (Table 1). The tree density decreased from RF into the exterior while sapling density was higher in nAL than in the other locations (Table 1 and Figure 3). In SC, the stem density in all categories showed a significant (ANOVA, $P < 0.05$) difference. The tree and seedling densities were higher in RF than in the exterior while the lowest density of saplings was found in dAL (Table 1). In AC, an effect of cleared forest edge was found. The tree, sapling and seedling categories had significant (ANOVA, $P < 0.05$) differences between RF and the exterior, with higher values found in RF than in the exterior (Table 1 and Figure 3).

Table 1 Vegetation structure, diversity and environmental parameters for three 20 × 150 m plots in three forest edge sites in lower montane forest in Um Phang Wildlife Sanctuary, western Thailand (values are mean ± SD).

| Plot site ^a | RF | | | nAL | | | dAL | | |
|--|--------------------------|---------------------------|--------------------------|---------------------------|--------------------------|---------------------------|---------------------------|--------------------------|--------------------------|
| | AB | AC | SC | AB | AC | SC | AB | AC | SC |
| Maximum canopy height (m) | 20 | 19 | 17 | 18 ^b | - | 17 | 13 | - | 6 |
| Stem density ^{c#1} | | | | | | | | | |
| seedlings (per 0.0001 ha) | 3.37 ^A ±3.24 | 6.73 ^B ±7.34 | 9.27 ^C ±6.06 | 1.80 ^A ±4.36 | 0.17 ^A ±0.59 | 1.07 ^A ±1.99 | 1.87 ^A ±1.92 | 0.97 ^B ±1.88 | 3.63 ^A ±3.99 |
| saplings (per 0.0016 ha) | 1.07 ^A ±1.36 | 2.07 ^A ±2.21 | 4.17 ^B ±3.24 | 4.20 ^A ±4.14 | 0.00 ^B ±0.00 | 2.76 ^A ±2.80 | 10.80 ^A ±11.43 | 0.67 ^B ±0.36 | 3.63 ^A ±3.99 |
| trees (per 0.01 ha) | 4.37 ^A ±2.61 | 2.20 ^B ±2.31 | 8.13 ^C ±4.73 | 2.17 ^A ±2.17 | 0.00 ^B ±0.00 | 0.07 ^B ±0.25 | 1.40 ^A ±1.49 | 3.07 ^A ±5.21 | 1.33 ^A ±2.69 |
| Stem density ^{c#2} | | | | | | | | | |
| seedlings (per 0.0001 ha) | 3.37 ^A ±3.24 | 6.73 ^A ±7.34 | 9.27 ^A ±6.06 | 3.37 ^A ±3.24 | 0.97 ^B ±1.88 | 3.03 ^B ±2.97 | 1.80 ^A ±4.36 | 0.17 ^B ±0.59 | 1.07 ^B ±1.99 |
| saplings (per 0.0016 ha) | 1.07 ^A ±1.36 | 2.07 ^A ±2.21 | 4.17 ^A ±3.24 | 10.80 ^B ±11.43 | 0.07 ^B ±0.36 | 3.63 ^A ±3.99 | 4.20 ^A ±4.14 | 0.00 ^B ±0.00 | 1.57 ^B ±2.80 |
| trees (per 0.01 ha) | 4.37 ^A ±2.61 | 2.20 ^A ±2.31 | 8.13 ^A ±4.73 | 1.40 ^B ±1.50 | 1.07 ^B ±1.21 | 1.33 ^B ±2.69 | 2.17 ^B ±2.17 | 1.26 ^B ±3.50 | 0.07 ^B ±0.25 |
| Basal area (m ² .ha ⁻¹) | | | | | | | | | |
| stem ≥ 5 cm DBH | 21.93 ^A ±2.23 | 9.67 ^A ±3.68 | 14.99 ^A ±6.77 | 6.94 ^B ±0.56 | 0.00 ^B ±0.00 | 3.38 ^B ±4.00 | 1.92 ^C ±11.96 | 0.00 ^B ±0.00 | 0.59 ^C ±0.74 |
| stem ≥ 10 cm DBH | 21.39 ^A ±2.14 | 8.55 ^A ±3.86 | 11.92 ^A ±6.89 | 6.75 ^B ±0.58 | 0.00 ^B ±0.00 | 2.97 ^B ±3.97 | 1.05 ^C ±2.39 | 0.00 ^B ±0.00 | 0.59 ^C ±0.70 |
| Shannon-Wiener index ^d | 2.52 ^A ±0.27 | 2.20 ^A ±0.12 | 2.40 ^A ±0.16 | 1.40 ^B ±0.34 | 0.73 ^B ±0.40 | 1.07 ^B ±0.40 | 1.58 ^B ±0.07 | 0.00 ^C ±0.00 | 0.52 ^C ±0.113 |
| RLI (%) | 20.19 ^A ±5.73 | 34.83 ^A ±10.05 | 25.11 ^A ±9.01 | 29.85 ^B ±3.65 | 94.95 ^B ±3.75 | 51.38 ^B ±10.10 | 64.15 ^C ±14.64 | 99.55 ^C ±0.75 | 83.33 ^C ±7.78 |
| SMC (%) | 43.04 ^A ±5.84 | 29.82 ^A ±4.09 | 30.67 ^A ±3.57 | 33.17 ^B ±4.84 | 24.11 ^B ±2.86 | 27.32 ^B ±3.95 | 21.81 ^C ±2.27 | 15.16 ^C ±2.42 | 22.47 ^C ±3.47 |
| SDb (g.cm ⁻³) | 0.73 ^A ±0.08 | 0.74 ^A ±0.07 | 0.83 ^A ±0.09 | 0.91 ^B ±0.09 | 0.86 ^B ±0.11 | 0.91 ^B ±0.09 | 1.06 ^C ±0.07 | 1.28 ^C ±0.10 | 1.15 ^C ±0.10 |

^a Site codes: AB = abandoned area, AC = annual cropping and SC = shifting cultivation.

^b Maximum canopy height values corresponding to remnant isolated trees in each site.

^c Stem size: seedlings (height < 50 cm), saplings (height < 1.30 m; DBH < 5 cm), trees (DBH ≥ 5 cm).

^d Shannon-Wiener diversity index (H') values as calculated for all tree species.

^{#1, #2} Symbols denote the comparison of stems within and among the three forest edge sites, respectively.

RLI = relative light intensity, SMC = soil moisture content, SDb = soil bulk density.

A,B,C Different letters denote values that are significantly ($P < 0.05$) different among the three forest edge sites.

The basal area was significantly (ANOVA, $P < 0.05$) different along the RF toward the exterior. The basal area for stems greater than or equal to 5 and 10 cm DBH in AB and SC of RF were significantly (ANOVA, $P < 0.05$) different in nAL and dAL (Table 1). Crown canopy tree

species such as *Castanopsis acuminatissima*, *Callophyllum polyanthum*, *Michelia floribunda* and *Choerospondias axillaris* had a large basal area only in the RF. Subcanopy and understorey tree species were most abundant but differed between land use practices and included *Styrax benzoides*,

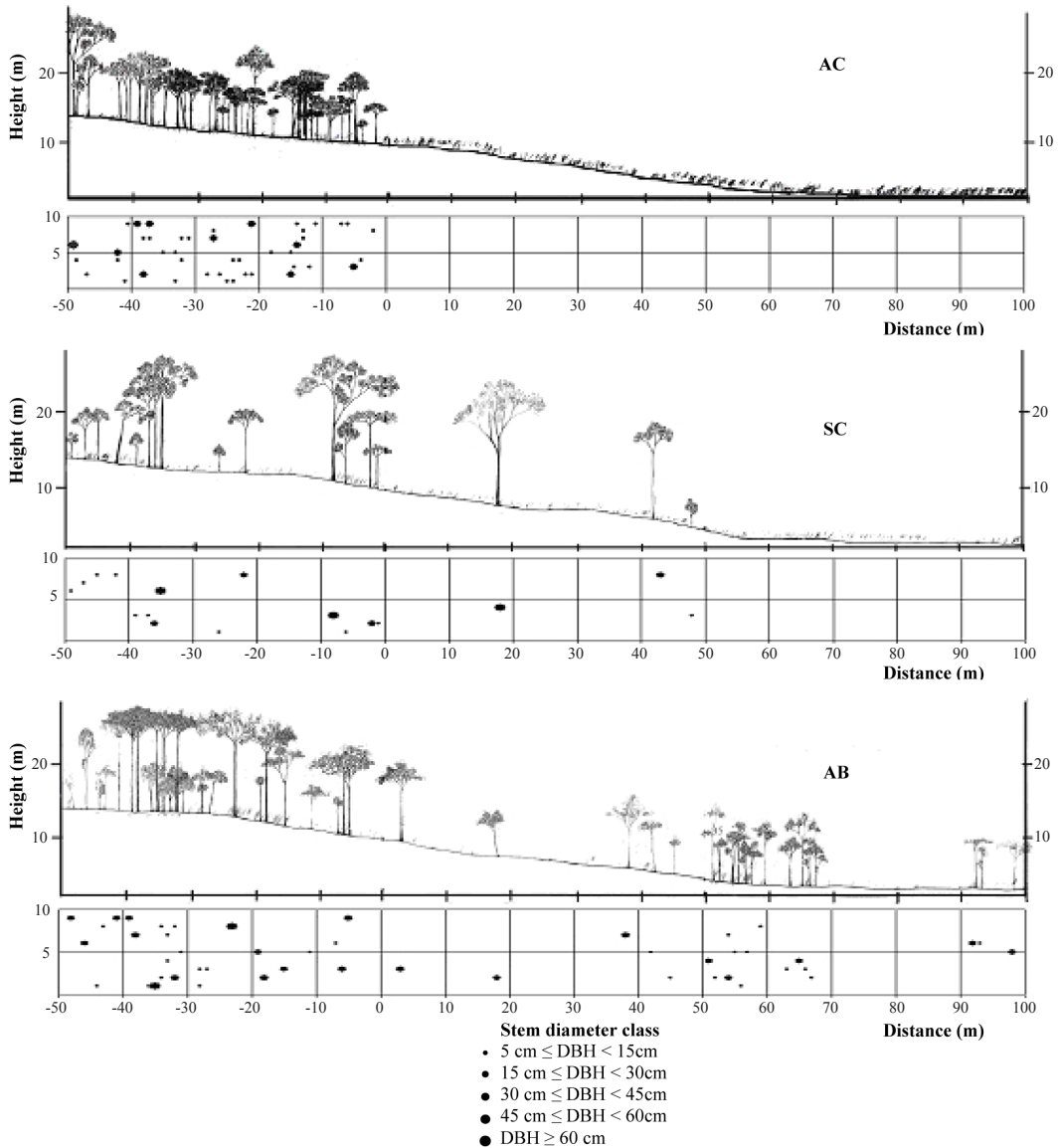


Figure 2 Schematic profile diagrams and horizontal stem distributions (DBH = diameter at 1.3 m above ground) in belt transects of 10 × 150 m along gradient distant from the remnant forest into near-to-edge and distant-from-edge corresponding to three agriculture land types, abandoned area (AB), annual cropping (AC) and shifting cultivation (SC).

Betula alnoides, *Diospyros glandulosa*, *Schima wallichii*, *Walsura robusta*, *Wendlandia tinctoria*, *Elaeocarpus sphaericus*, *Eurya acuminata* and *Duabanga grandiflora* in the SC and AB, while, *Archidendron clypearia*, *Macaranga kurzii* and *Albizia chinensis* were the dominant species in AC.

Considering the diameter class distribution of tree over 10 stems from RF toward exterior, the species *Castanopsis acuminatissima*, *C. tribuloides*, *Michelia floribunda*, *Schima*

wallichii, *Styrax benzoides*, *Colona elobata* and *Eurya acuminata* succeeded in establishing in RF, which contained numerous trees from a small size (5–10 cm DBH) to large sizes (over 30cm DBH, Figure 4); however, only *Castanopsis acuminatissima* and *Schima wallichii* had a higher abundance in nAL than the other species. In contrast, *Rhus chinensis* had high abundance in the small size class (5–10 cm DBH) in the exterior and a few were found in RF (Figure 4).

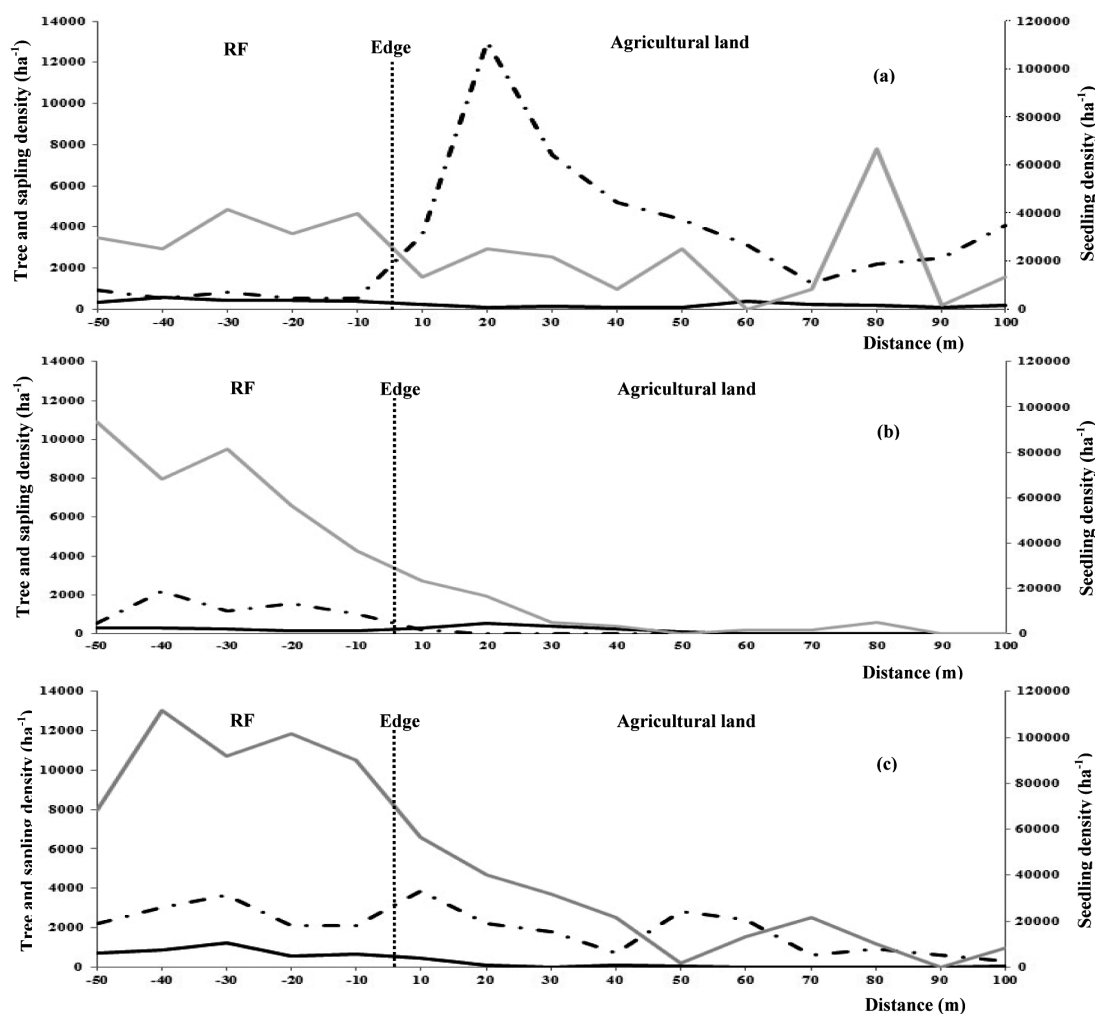


Figure 3 Distribution of stem numbers along a LMF interior-edge-exterior distance gradient for three different land uses; (a) abandoned area; (b) annual cropping; (c) shifting cultivation. Black line, dashed black line and grey line indicate tree, sapling and seedling classes, respectively.

Tree diversity determined using the Shannon-Weiner index (H') was fitted to a polynomial function for all sites, and showed a decreasing trend from RF into the exterior (Figure 5). In AB land uses, the tree diversity showed a significant (ANOVA, $P < 0.05$) difference only

between RF and the exterior; however, in the AC and SC land uses, a significant (ANOVA, $P < 0.05$) difference among the three forest edges was detected, with a higher diversity found in RF followed by nAL and dAL, respectively (Table 1).

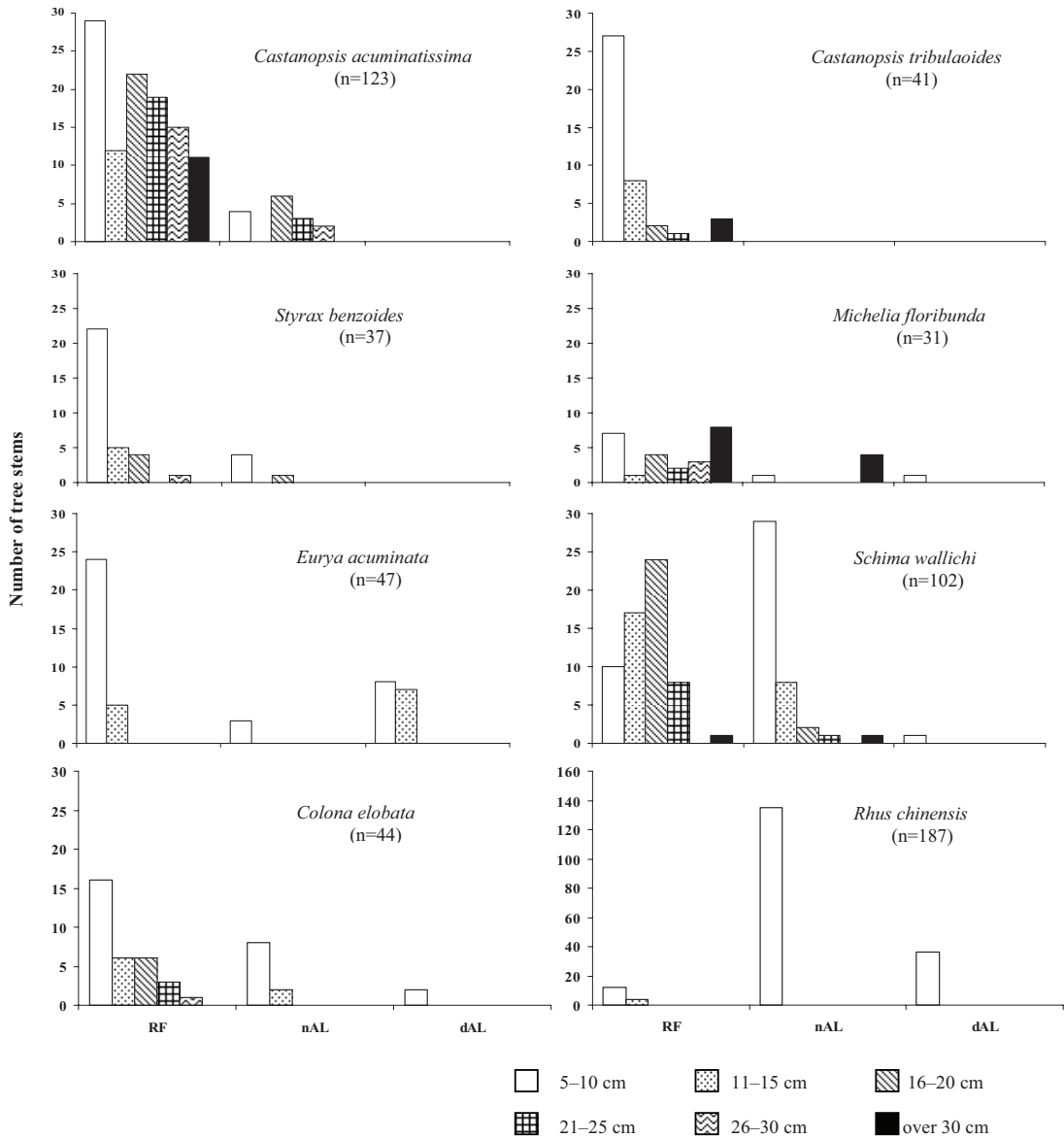


Figure 4 Diameter class distribution of tree species (with more than 10 stems) across all three sites distributed from remnant forest (RF), near-to-edge (nAL) and distant-from-edge (dAL) of agricultural lands.

Species traits along the forest edge

The different land use practices, AB, SC and AC, had an effect on RLI, SMC and SDb. RLI was significantly (ANOVA, $P < 0.05$) different among the forest edges in AB and SC. A higher value was found in dAL followed by nAL and RF, respectively (Table 1). In AC, RLI had a very high intensity (over 90%) in the exterior and was significantly (ANOVA, $P < 0.05$) different from RF (Table 1). SMC and SDb were significantly (ANOVA, $P < 0.05$) different in all land use practices. SMC decreased from RF into nAL and dAL, respectively, contrasting with the trend for SDb.

The CCA ordination analysis of the 'tree' category found that the species-environment correlation had a significant ($P < 0.05$) correlation. RLI and SDb showed a negative correlation between Axis1 ($r^2 = 0.83$ and 0.79 , respectively). In contrast, SMC had a positive correlation with

Axis1 ($r^2 = 0.83$). The species traits to species-environment correlation according to CCA analysis could be divided into three groups (Figure 6). The first group was species that responded well to the high variation in RLI and SDb such as *Rhus chinensis* (RHUSCHI), *Erythrina subumbrans* (ERYTHSU) and *Albizia chinensis* (ALBIZCH). This group can be classified as pioneer species due to their light-demanding trait. The second group consisted of species with a preference for high SMC such as *Castanopsis acuminatissima* (CASTAAC), *C. tribuloides* (CASTATR), *Lithocarpus grandifolius* (LITHOGR), *Michelia floribunda* (MICHEFL), *Diospyros glandulosa* (DIOSPGL) and *Styrax benzoides* (STYRAAP). Most of these species were dominant tree species of LMF. The third group consisted of species with a correlation intermediate to RLI and SMC, including *Shima wallichii* (SCHIMWA), *Eurya accuminata* (EURYAAC), *Wendlandia tinctoria* (WENDLTI),

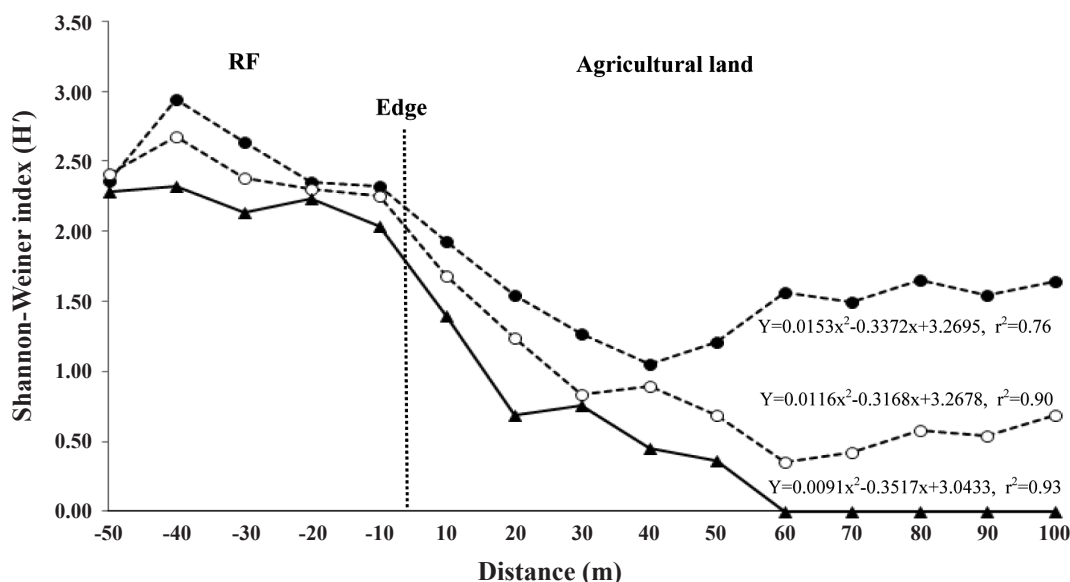


Figure 5 Tree diversity by Shannon-Weiner (H') index values for 10×10 m quadrats positioned along remnant forest (RF) toward the exterior gradient in degraded lower montane forest, western Thailand. Closed circle, open circle and triangular symbols indicate the diversity in three different land use types—abandoned area (AB), annual cropping (AC) and shifting cultivation (SC), respectively.

Colona elobata (COLONEL), *Broussonetia papyrifera* (BROUPSA), *Macaranga kurzii* (MACARKU), *Glochidion kerii* (GLOCHKM), *Cordia cochinchinensis* (CORDICO) and *Artocarpus dadah* (ARTOCDA), among others.

This species group may be classified as forest edge species due to its position on the environmental gradient.

On the basis of the relative abundance of stems (including the tree and sapling categories),

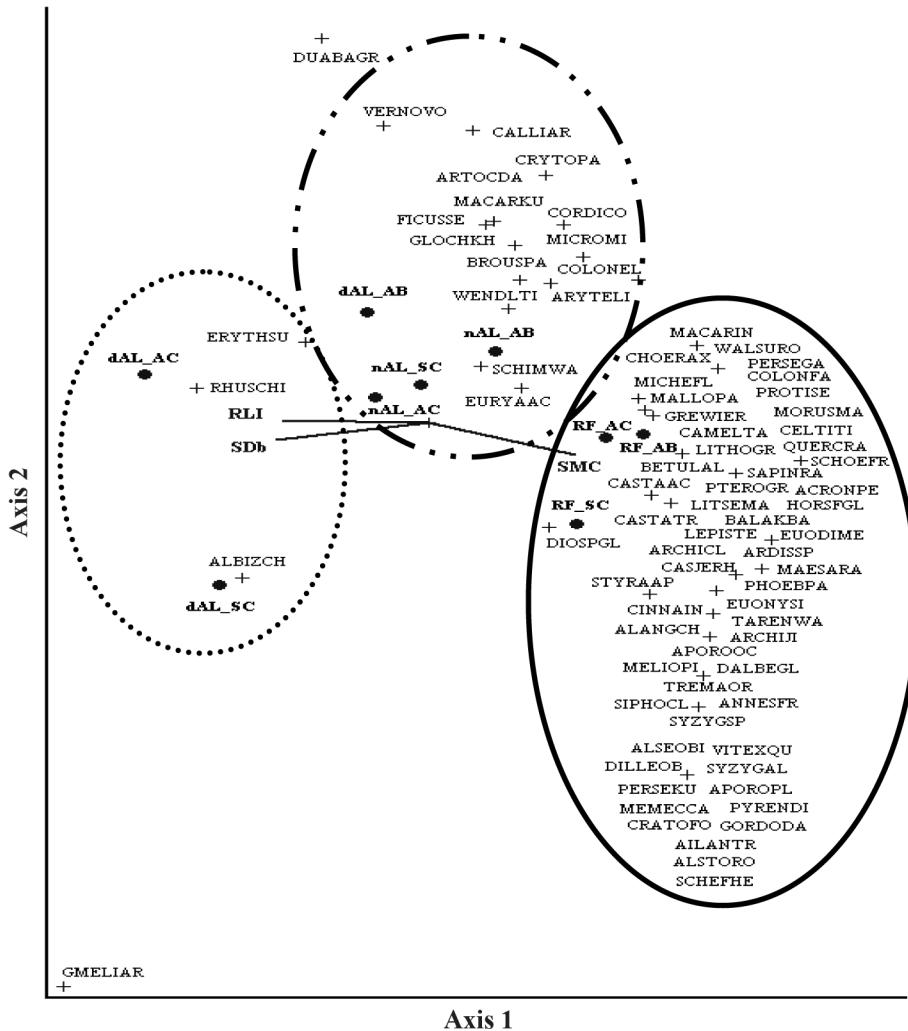


Figure 6 Two-dimensional simplified canonical correspondence analysis ordination of woody trees and shrubs (capital letter abbreviations, see Appendix A) assemblages on environmental gradients (RLI = relative light intensity, SMC = soil moisture content, SDb = soil bulk density) in degraded lower montane forest. Closed dots represent sites distributed from remnant forest into near-to-edge (nAL) and distant-from-edge (dAL) of three different land use types—abandoned area (AB), annual cropping (AC) and shifting cultivation (SC), while (+) symbols indicate the position of species against Axis 1 and Axis 2. The three circular line types indicate the different relationships between plant groups and environmental factors.

a distribution map along the forest edges was developed encompassing a canopy species (*Castanopsis acuminatissima*), a shade-tolerant species (*Shima wallichii*) and a pioneer species (*Rhus chinensis*). In all practices, the abundance

of *Castanopsis acuminatissima* gradually decreased from RF into the exterior, especially in AC. *Rhus chinensis* had higher abundance in the exterior than RF, especially in nAL in which a high abundance peak was detected (Figure 7). However, there

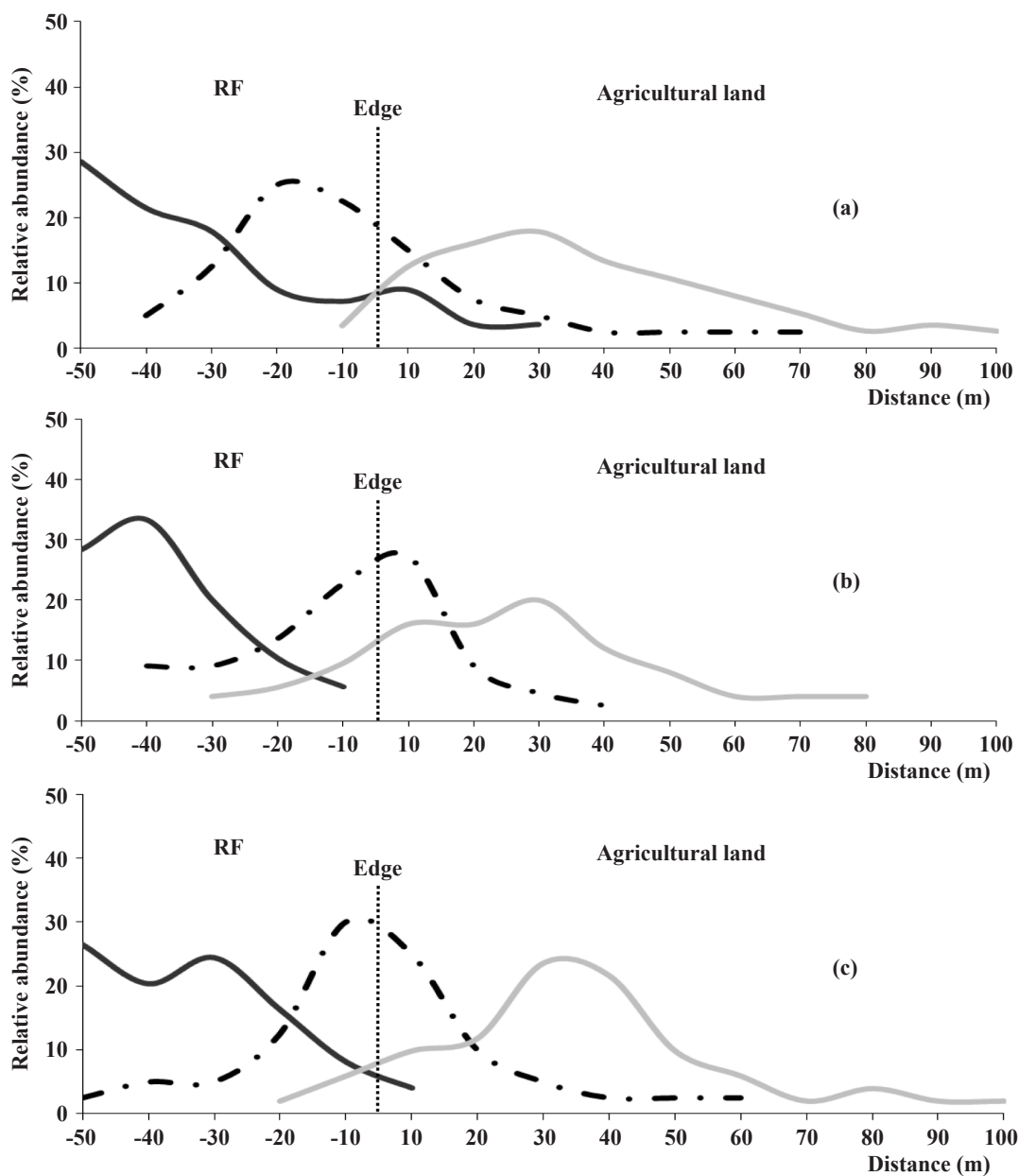


Figure 7 Distribution map of spatio-temporal forest edge succession including *Castanopsis acuminatissima* (black line), *Shima wallichii* (dashed black line) and a *Rhus chinensis* (grey line) distributed in (a) abandoned area, (b) annual cropping and (c) shifting cultivation.

was evidence of some invasion into RF. Some individuals of *Shima wallichii* were simultaneously distributed from RF into the exterior; however, the highest abundance was found close to the edge (-10m to 0m) for all practices (Figure 7).

DISCUSSION

Forest-edge dynamics

Forest edges are usually characterized by strong environmental changes, especially higher air temperature and lower relative humidity and soil moisture than in the forest interior (Kapos, 1989; Matlack, 1993; Young and Mitchell, 1994; Chen *et al.*, 1995). The light transmittance pattern (RLI) was from RF into nAL and dAL. In contrast, SMC had a different tendency which decreased from RF into nAL and dAL. The strong RLI, due to light exposure at the forest exterior, could stimulate seed germination and the growth of pioneer species along the forest edge (Ranney *et al.*, 1981; Lovejoy *et al.*, 1986; Aide and Cavelier, 1994; Marod *et al.*, 2004). This may initiate a natural succession process, from grassland towards mature forest (Kappelle *et al.*, 1996) and an increase in pioneer species, in addition to the RF which may result in higher species richness along the edge. Gradual changes in vegetation have been observed from the RF edge bordering agricultural land (Pattanavibool and Dearden, 2002; Fukushima *et al.*, 2008). The vegetation structure and species composition along the RF edge gradient have been reported to change from an assemblage of high density canopy and subcanopy trees in RF, decreasing into open areas and less dense stands at the edge, with a slight increase in the small sapling density in abandoned fields (Laurance *et al.*, 1998; Sizer and Tanner, 1999). Pioneer species, *Rhus chinensis* and *Chromolaena odorata*, occupied the exterior in the early successional stage, while canopy tree species such as *Castanopsis acuminatissima*, *C. tribuloides* and *Lithocarpus grandifolius*, among others were only present inside RF rather than in the exterior.

This was likely caused by the high level of tree mortality (Laurance *et al.*, 2002; Nascimento and Laurance, 2004), as a result of the influence of land use practices (Dearden, 1996; Pattanavibool and Dearden, 2002). This pattern was indicated by the higher mean basal area of trees over 10 cm DBH concentrated in RF rather than in nAL and dAL (Table 1), with the highest value being found in the AB followed by SC and AC, respectively. The later successional species were characterized by *Shima wallichii*, *Eurya acuminata*, *Broussonetia papyrifera*, *Macaranga kurzii* and *Colona elobata*, among others. These species were classified as forest edge species which are known to regenerate from gap and ecotone areas in the LMF (Schmidt-Vogt, 1998; Hara, 2004; Kubota *et al.*, 2005; Fukushima *et al.*, 2008) and so may buffer the remaining mature forest core and simultaneously induce forest recovery processes in adjacent abandoned agriculture areas along the LMF edge (Kappelle *et al.*, 1994). This suggests that the edge effect is a major factor that not only changes the climatic conditions but also impacts the vegetation communities in the RF (Laurance *et al.*, 2002).

The Shannon-Weiner diversity index also displayed the same trend as basal area, decreasing from RF into the edge-exterior. The diversity varied among the land use practices with a significantly higher value found in AB compared to SC and AC, respectively (Figure 5). Thus, the vegetation succession along the forest edge was influenced by both human activities and agricultural systems with a higher effect found in the intensive/permanent cultivation than in AB and SC (Schmidt-Vogt, 2001). This type of cultivation may induce an inappropriate environment for tree regeneration to a greater extent than other systems. In addition, there were indications of changes in the woody and herbaceous community structure from a few meters to 20–30 m from the border (Matlack, 1993; Gehlhausen *et al.*, 2000; Honnay *et al.*, 2002), depending on the edge type. Therefore, perennial herbs, such as *Chromolaena*

odorata, persist much longer as a pioneer species under AC than under SC and AB land uses, especially, on fallow land (Cagnolo and Valladares, 2006). Farmers do not leave relict emergent trees on their fallow land (Kunstadter, 1978; Anderson, 1993; Schmidt-Vogt, 1998) and this may prohibit vegetation succession along the forest edge. The differences in edge type determined by disturbance frequency helped explain some of the between-site variation in the environment and the species composition.

Implications for forest restoration

One alternative approach for forest conservation by the restoration of native forest is to accelerate regeneration by assisting the natural succession processes (Hardwick *et al.*, 1997; Laurance *et al.*, 2006). The main objective of this approach would be to increase the recruitment of pioneer and late-pioneer species along the forest edge (Laurance *et al.*, 1998). The importance of basing tropical forest restoration on an understanding of ecological processes is often emphasized (Lugo, 1988; Janzen and Vazquez-Yanes, 1991) in order to determine the specific factors that limit regeneration in deforested areas. In practice, farmers have often destroyed the remnant trees and seeds through intensive agriculture, so forest regeneration in abandoned agricultural clearings usually depend on seeds dispersed from the forest (Guevara *et al.*, 1986; Nepstad *et al.*, 1990). The present study showed that *Shima wallichii* and *Rhus chinensis* had very high abundance and became the dominant species in the exterior which may have been supported by their seed dispersal after slash-and-burn activity (Schmidt-Vogt, 2001; Elliott *et al.*, 2003; Hara, 2004; Ruankaew, 2004). This result may have both positive and negative effects on the vegetation structure and species diversity along the forest edge, by changing the external physical and biotic factors. Species of *Shima wallichii* have a small winged seed and are widely dispersed

by wind (Appendix A), which may also promote their high seedling germination and rapid spread in successional forest stands (Fukushima *et al.*, 2008), while the small seeds of *Rhus chinensis* were dispersed by birds and wind (Appendix A). On the other hand, canopy tree species such as *Castanopsis accuminatissima* (Fagaceae) were distributed only in nAL of AB, indicating the potential impact of the disturbance frequency on tree regeneration (Schmidt-Vogt 1998), especially for shade-tolerant species. The germination of seedlings of *Castanopsis accuminatissima* might be limited at some sites because the fruit (usually a nut) of Fagaceae are relatively large and are only dispersed over relatively short distances by animals, though some seeds can still be found distant from the edge (Holl, 1999). However, wildlife abundance, especially of large frugivorous mammals and birds that can act as the vector for the dispersal of a wide range of seeds into clearings, also plays a significant role in promoting secondary growth along the forest edges (Corlett, 1996; Noma and Yumoto, 1997; Barlow *et al.*, 2006; Marod *et al.*, 2010).

CONCLUSION

The vegetation structure and floristic composition along the edge of montane forest were strongly impacted on by different agricultural land practices. Where there was intensive land use, only low herbaceous vegetation was found, while areas subjected to shifting cultivation and abandoned farmland showed the presence of patches occupied by secondary shrubs and trees that were recovering to secondary forest. Different practices measured by their disturbance frequency accounted for certain vegetation structures and floristic compositions between sites and also had an influence on environmental changes, especially the relative light intensity which increased from RF into nAL and dAL.

The present study identified some of

the main characteristic plant species indicative of edge conditions. It is expected that spatio-temporal changes at the RF edge will include the assemblage of secondary vegetation in the course of time and leveling of the forest-agricultural-land border as land use intensity is reduced. Forest edge species will help buffer the remaining mature forest core and simultaneously induce forest recovery processes in the abandoned areas further away to develop a more diffuse edge. It is recommended that most typical forest edge species such as *Shima wallichii* and *Eurya acuminata* and also crown canopy trees such as *Castanopsis accuminatissima*, *Michelia floribunda* and *Choerospondias axillaris* should be included in experimental programs where the objectives are the ecological restoration of disturbed lower montane forest. Simultaneously, research should be directed at the natural history of these species, their role in forest regeneration and the microhabitat conditions they require for successful establishment. This knowledge will be crucially important to the maintenance and restoration of lower montane forest ecosystems in the long term.

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