



Bird conservation in extremely small tropical rainforest patches in southwest China

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ABSTRACT

Tropical forests support high levels of avian diversity but these areas have been impacted by extensive deforestation. The lowland tropical rainforest of Xishuangbanna in southwest China is one such impacted hotspot. In this region, rubber (*Hevea brasiliensis*) plantations have replaced most of the original forest, but numerous small fragments of forest remain. To assess the value of these fragments to avian diversity, we classified the fifteen selected remnant forest patches into three size categories, <1 ha, 1–3 ha, and 3–6 ha, and conducted avifauna surveys. We found that 97 bird species inhabit the fragments and that bird diversity increases with patch area. The resident species composition similarity index was low among forest patches, both within and across size categories, and also low between the wet and dry seasons. Ground insectivores, frugivores and large bird species were the most vulnerable to habitat fragmentation. Our study encouragingly shows that fragments can play an important role in bird conservation in regions that now lack large tracts of natural forest; however, current patches are too small to support stable annual populations of some species, while others are more tolerant of fragmentation. We discuss these findings and provide conservation strategies to improve bird conservation within rubber plantations in southern China.

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1. Introduction

Despite their extremely high biological diversity (Myers et al., 2000; Raven, 1981), tropical forests are being lost at an alarming rate (Brooks et al., 2002; FAO, 2006; Lambin et al., 2003). The region of tropical east Asia is of particular concern due to high relative rates of deforestation and low levels of remnant forest (Jepson et al., 2001; Laurance, 2007; Li et al., 2009; Sodhi et al., 2004a). More than half of the original forest cover in tropical east Asia has been cleared (Corlett, 2009), and even today some countries in this region still have a high rate of forest loss (Hansen et al., 2008). If this rate of habitat loss continues, it is estimated 13–42% of Southeast Asia's taxa will be wiped out by 2100 (Brook et al., 2003).

The negative impacts of natural forest-cover change are complex and broad (Lambin et al., 2003), one of the primary concerns is that changes in forest-cover will alter the biodiversity of the Earth (Castelletta et al., 2005; Sala et al., 2000; Sodhi et al., 2010). In tropical east Asia, for example, a large portion of original forest has been converted to cash crops, such as oil palm (*Elaeis guineensis*) and rubber (*Hevea brasiliensis*) (Koh and Wilcove,

2008; Fitzherbert et al., 2008; Li et al., 2007), leading to the extinction of many endemic species (Beukema et al., 2007; Najera and Simonetti, 2010; Sheldon et al., 2010). Tropical forests are the focus of current and future extinctions (Bradshaw et al., 2008); however, except for measurements of forest loss, our understanding of the diversity of tropical Asia lags behind that of other regions (Corlett, 2009; Sodhi and Liow, 2000).

The forests of the Xishuangbanna Dai Autonomous Prefecture are located at the southwestern margin of China and are renowned for their especially high biodiversity. The region accounts for 0.2% of the land area of China, but contains approximately 16% of China's higher plant species, 21.7% of its mammals, 36.2% of its birds, and 14.6% of its amphibians and reptiles (Zhang and Cao, 1995). New bird species for China continue to be recorded in this region (e.g. *Anthreptes malacensis*, Wu et al., 2010). Unfortunately, land cover change dynamics across Xishuangbanna are representative of changes across tropical Asia (Fox and Vogler, 2005). From 1976 to 2003, 67% of lowland tropical rainforest (below 800 m) was replaced by rubber plantations (Li et al., 2007). Although the region has now developed an extensive protected area system, with about 12% of the land area covered by nature reserves (Guo et al., 2002), it continues to undergo change from rubber plantation expansion and other economic activities, such as food and tea production. Any remaining biodiversity in these severely degraded lowland tropical habitats requires urgent protection.

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Some of the ecological consequences of forest fragmentation and degradation for flora dynamics have been studied in this region (Mo et al., 2011; Zhu et al., 2000, 2004), but the actual value of this fragmented habitat for native bird diversity remains unclear (Wang and Young, 2003; Yang et al., 1985). Here, we conduct bird surveys in remnant forest patches within rubber plantations with the aim of better understanding the prospects for forest bird conservation in human-modified environments. We first examine avian diversity across the fragments, then determine the patch sizes which are able to sustain a year-round stable bird community by comparing overall species composition between the wet and dry season. Comparisons with bird species composition in local reserves are used to determine which species are more vulnerable to fragmentation. Our results will guide direct conservation planning in this important region in China, and also inform land managers throughout tropical Asia struggling with conserving biodiversity within agricultural ecosystems.

2. Methods

2.1. Study area

This study was conducted across 6000 ha (centered at 21°53'N, 101°16'E) within Xishuangbanna, Yunnan province, China. This region is part of tropical east Asia (Corlett, 2009) and borders Laos to the south and Myanmar to the southwest (Fig. 1). The climate is characterized by two distinct seasons: dry and relatively cool from November to April and wet and hot from May to October. The main forest type in the study area changes from tropical rainforest to subtropical evergreen broad-leaved forest with elevation (Wu et al., 1987). Areas suitable for rubber trees are less than 800 m above sea level and often covered by tropical rainforest. From 1976 to 2003 the total number of forest fragments in Xishuangbanna increased by 50% and mean patch area decreased by 60% to an average of 18 ha in 2003 (Li et al., 2009). Excluding larger forest patches in nature reserves and in the upland region, the average area of forest patches within the lowland rubber plantations is much less than 18 ha.

In the study area, plant species diversity and composition has changed with fragmentation. For example, the once dominant species *Barringtonia macrostachya* disappeared from fragmented forests in our study area, and *Antiaris toxicaria* now dominates. Liana and microphanerophyte species have increased, but epiphyte, megaphanerophyte, mesophanerophyte and chamaephyte species have declined (Zhu et al., 2004).

2.2. Bird surveys

Fifteen tropical forest patches (Appendix 1) within the rubber plantation matrix were selected randomly for bird surveys. By employing QuickBird satellite images (2006, spatial resolution 0.6 m) and ArcGIS 9 software (ESRI 2004) we calculated the area of each fragment. We then classified forest patches into three categories: <1 ha ($n = 6$), 1–3 ha ($n = 5$), and 3–6 ha ($n = 4$). All patches were completely surrounded by rubber plantations (Fig. 1). In our study site, the satellite images in 2006 are applicable for calculating patch area in 2010/2011 because the forest patches are located on steep slopes and have not suffered further fragmentation since the mid-1990s.

Regular bird sampling was conducted from May to October 2010 (wet season), and November 2010 to April 2011 (dry season). All forest patches were visited during each 10–15 d sampling period, and 12 counts (a count is defined as a survey of all the patches over a 10–15 d sampling period) were made in total. We employed the point count method (Bibby et al., 2000) to survey bird communities. The number of sampling points in each fragment was determined according to the area of the fragment, which was one point in fragments less than 1 ha, two points for 1–3 ha fragments and three points for 3–6 ha fragments. In total, 26 points were employed and each point was repeated 12 times during surveys from 2010 to 2011. Counts were conducted during times of high bird activity from 0800 to 1100 h and 1500–1800 h in good weather (no heavy rain, strong winds or thick fog). During surveys we recorded bird species and the number of individuals seen or heard within a 50 m radius from a fixed point, where some trees acted as the boundary. The time duration at each point was 10 min. Bird recording began 1–2 min after we reached the survey point to allow bird activity to resume. Birds flying over the surveying point were not taken into account. In the smaller forest patches (<1 ha), survey points were randomly placed along trails; in larger forest patches (1–3 ha and 3–6 ha), successive points were spaced at least 200 m apart for independence (Gregory et al., 2004). The order of counting the points was regularly rotated to minimize possible surveying order and time biases on bird presence and absence at each point. To determine whether our sampling was enough to represent the bird community in the fragment, we drew sampling effort curves for each category of patch using only resident species because they were the most representative species for the local bird community. An asymptote was reached suggesting that adequate sampling was achieved (Fig. 2).



Fig. 1. Location of study site (left) and photographs of three forest fragments in a rubber plantation landscape (right). Circles are forest patches used for bird surveying. All patches used for bird surveying were completely surrounded by rubber plantation.

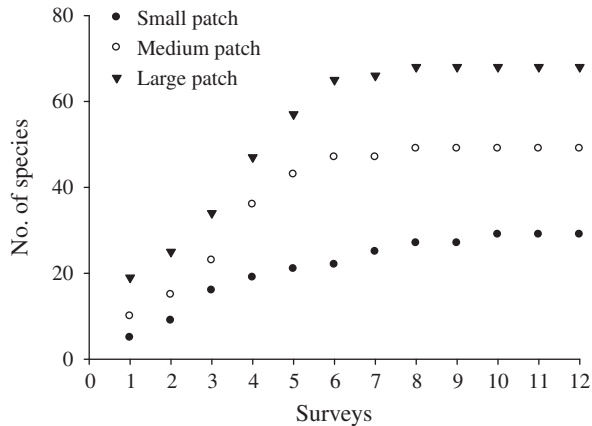


Fig. 2. Cumulative number of species recorded during point surveys in different forest patches.

2.3. Species classification

Birds were classified into residents or migrants based on whether they occur throughout the year or use this region only for wintering or breeding (Yang, 2004; Yang et al., 1995). To determine whether fragmentation influences foraging guilds differently we classified birds into frugivores, insectivores, nectarivores, and omnivores. We also classified species as large (>40 cm), medium (20–40 cm), and small (<20 cm) by body length to determine which size is more vulnerable to fragmentation. Feeding guilds and body sizes were based on the Avifauna of Yunnan (Yang, 2004; Yang et al., 1995) and field guides (MacKinnon et al., 2000).

2.4. Data from nature reserves

To assess the effect of forest fragmentation, we compared presence/absence data from our surveys to the species list from the Xishuangbanna National Nature Reserve, which is within about 5–10 km of the patches surveyed in this study. The total area of the nature reserve is about 92,683 ha and the diversity in these areas has been protected since the 1950s. Given the proximity and known history of the fragments, it is a reasonable assumption that their original species composition was similar to that in the nearby nature reserve. We only compared the presence/absence data and not the more complete estimates of abundance because the survey methods employed in the nature reserve (including variable distance line-transect walks, mist netting, as well as interviews with local hunters by showing them pictures; Xishuangbanna National Nature Reserve, 2006) were different with those used in this study. However, both surveys in reserves and fragments are focused on bird lists, and this gave us the opportunity to examine differences in bird presence/absence data between reserves and fragments. Migrants were excluded from all analyses. Avifauna of the nature reserve was derived from Xishuangbanna National Nature Reserve (2006).

2.5. Data analyses

Spearman correlations were used to calculate the relationship between the avian community (number of species and number of individuals) and forest patch area. Avian richness and abundance were indicated as 'number of resident bird species/point' and 'number of resident individuals/point'. Student T tests were used to compare avian richness and abundance between the dry season and wet season. We used ANOVA to check for differences in bird abundance (classified by feeding guild and body size) both within

and across different categories of forest patch area. Chi-square tests were employed to compare the composition of species richness in forest patches to nature reserves.

The Jaccard similarity index was used to analyze the similarity in species composition for resident species between any two forest patches (Boyle et al., 1990). We then summarized the similarity index based on different categories of forest patch (smaller, medium and large). To understand whether the current area of a forest patch can support a year-round stable bird community we divided the surveys into dry and wet seasons. The Jaccard similarity index was also used to calculate the similarity in species composition between wet and dry seasons. We summarized this by forest patch area (small, medium and large) and used T-tests to look for differences.

3. Results

3.1. Community composition in the rubber-dominated fragment

A total of 97 bird species and 3642 individuals were detected across 15 patches, ranging in area 0.6–6.5 ha. Seventy-three species were residents and 24 were migrants (including winter migrants and summer migrants). Sixty-five species were small, 27 were medium, and five were large. Eight species were frugivorous, 72 were insectivorous, 10 were nectarivorous and seven were omnivores. Three species were listed as vulnerable in China (Appendix 2). The total number of species and individuals present in each patch increased with patch area (Spearman correlation, species: $r^2 = 0.85$, $n = 15$, $p < 0.001$; individuals: $r^2 = 0.91$, $n = 15$, $p < 0.001$, Fig. 3). Only four species were recorded in all patches: the common tailorbird (*Orthotomus sutorius*), sooty-headed bulbul (*Pycnonotus aurigaster*), Japanese white-eye (*Zosterops japonicus*) and crimson sunbird (*Aethopyga siparaja*).

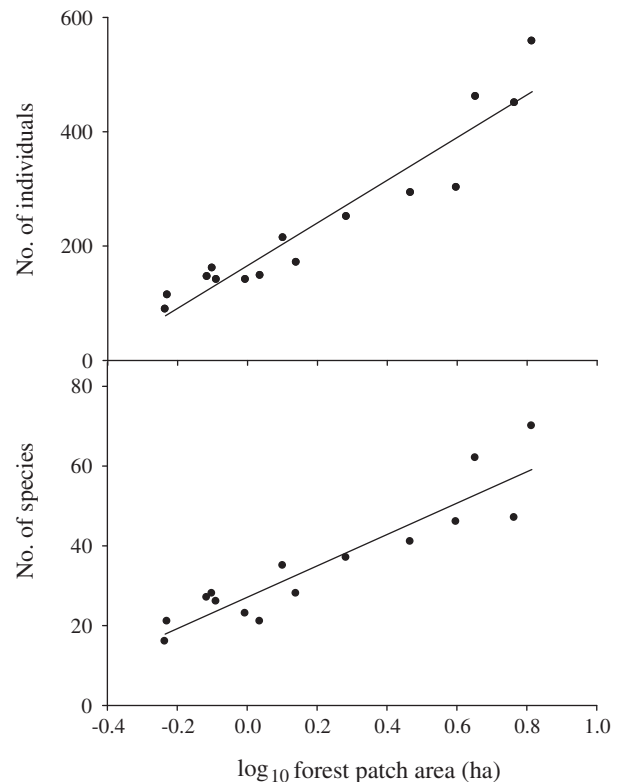


Fig. 3. Regression between number of individuals, species and forest patch area.

3.2. Similarity in species composition

Large patches (3–6 ha) harbored 46–70 species (56.3 ± 11.7). The mean proportion of species in large fragments present in smaller fragments increased with patch area from 0.42 (± 0.10 SE) in fragments <1 ha to 0.57 (± 0.14) in fragments 1–3 ha. Although 97 bird species were observed, the similarity index for resident species composition (73) was very low both within (Fig. 4A) and between (Fig. 4B) the three sizes of forest patch, suggesting that there are very large differences in species composition among different patches. Even in the relatively large patches (3–6 ha), the percentages of species shared were also very low (0.35 ± 0.01). To understand which patch size is capable of sustaining a year-round

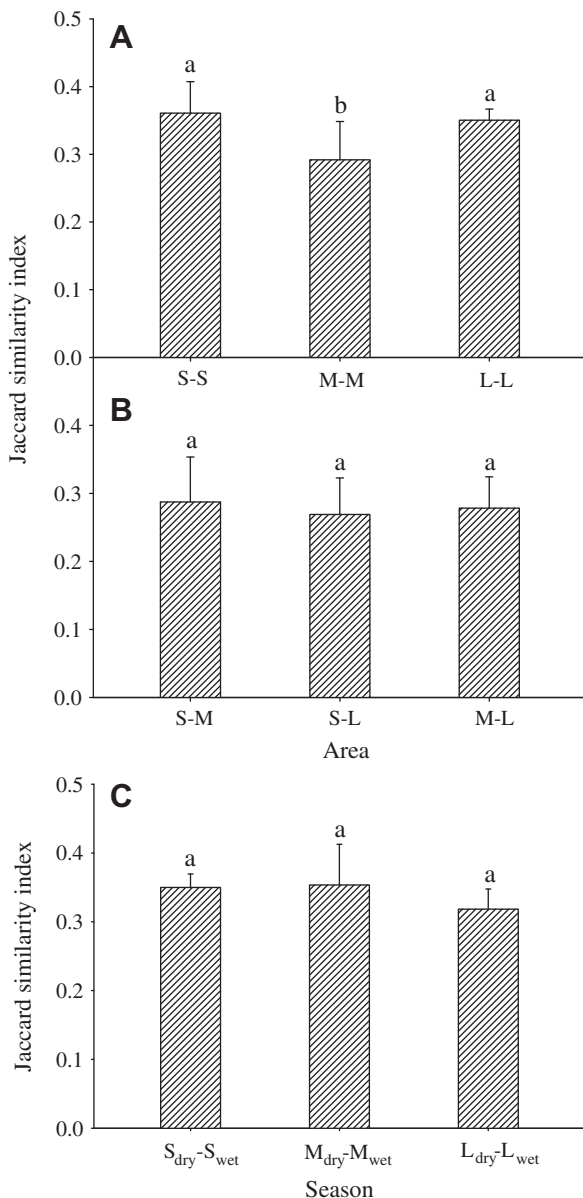


Fig. 4. Jaccard similarity index in species composition in (A) Similarity indices for small–small patch (S–S), medium–medium patch (M–M), and large–large patch comparisons (L–L); (B) similarity indices for fragment of small–large comparison (S–L), small–medium comparison (S–M) and medium–large (M–L) comparison; (C) similarity indices between dry and wet seasons for small_{dry}–small_{wet} (S_{dry}–S_{wet}), medium_{dry}–medium_{wet} (M_{dry}–M_{wet}), and large_{dry}–large_{wet} (L_{dry}–L_{wet}) comparisons. Within each panel, bars with the same letter are not significantly different at $p < 0.05$ (ANOVA).

stable community, we calculated a similarity index for each small_{wet}–small_{dry}, medium_{wet}–medium_{dry}, and large_{wet}–large_{dry} comparison to analyze the similarity between the dry and wet seasons. We found that the avian community was unstable over a year, as only about 30% of species were recorded in both the dry and wet seasons (Fig. 4C).

All 29 species found in small patches were also present in medium and large patches. Five of the 49 species found in medium patches were absent from small and large patches. Although 70 species were found in larger patches, 20 of these were unique to large patches (Appendix 2). Most of the unique species in larger patches were ground insectivores and were observed in the interior zones of the large patches only.

3.3. Seasonal effects on species richness and abundance

During the dry season, 73 species (1877 individuals) were observed, 32 of which were unique to the dry season. Forty-one species were present in the wet season (1246 individuals) but none were unique to the wet season. Species richness (number of birds per point observation) and abundance (number of individuals per point observation) was significantly higher in the dry season than in the wet season (richness, $t = 13.65$, $df = 298$, $p < 0.001$; abundance, $t = 10.82$, $df = 298$, $p < 0.001$) (Fig. 5).

3.4. Effect of patch size on avian feeding guilds and size

Insectivorous birds were dominant in terms of the average number of bird species and individuals observed per survey point observation (Table 1). Associated with increasing patch area, the richness of insectivorous, frugivorous and nectarivorous birds also increased; however, the abundance of frugivorous and nectarivorous birds did not increase with patch size. The richness and abundance of omnivorous birds were less dependent on patch size (Fig. 6). Small species dominated small, medium and large forest patches (Table 2). The richness of small and medium sized species increased significantly with patch area; however, the abundance of medium sized species did not increase with patch area. Both the richness and abundance of large species were not affected by patch size (Fig. 7).

3.5. Patch communities compared to nature reserves

The total number of resident species in the fragments was low: 73 of the 265 species found in nature reserves were recorded in

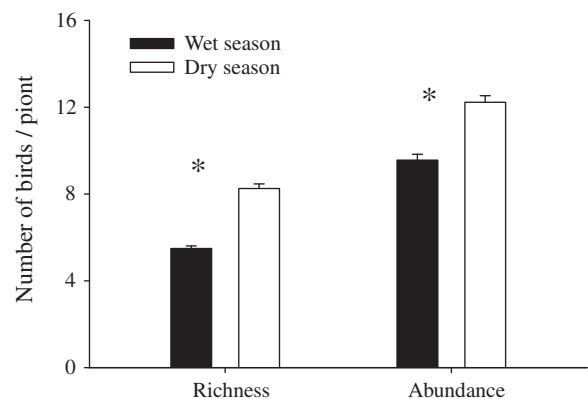


Fig. 5. Species richness (birds/point observation) and abundance (individuals/point observation) in the dry (white bars) and wet season (black bars). (Richness, dry = 9.56 ± 0.27 se; wet = 5.49 ± 0.12 . Abundance, dry = 12.23 ± 0.30 ; wet = 8.25 ± 0.21 . $N = 150$ for each; *: significant at $p < 0.01$).

Table 1
Bird richness and abundance by feeding guild across different patch areas in a rubber plantation.

Patch area ^a	Feeding guilds								ANOVA ^d
	Nectarivore		Frugivore		Insectivore		Omnivore		
	Rich. ^b	Abun. ^c	Rich.	Abun.	Rich.	Abun.	Rich.	Abun.	
Small	1.08 ± 0.10	1.56 ± 0.17	0.82 ± 0.07	2.14 ± 0.21	2.2 ± 0.11	4.06 ± 0.24	0.74 ± 0.06	1.96 ± 0.22	$F_{rich:3,284} = 57.7$; $F_{abun:3,284} = 28.2$
Medium	1.31 ± 0.09	1.99 ± 0.16	0.99 ± 0.07	2.27 ± 0.18	2.2 ± 0.10	4.59 ± 0.18	1.00 ± 0.08	1.68 ± 0.16	$F_{rich:3,380} = 42.9$; $F_{abun:3,380} = 64.2$
Large	1.58 ± 0.07	1.77 ± 0.08	1.92 ± 0.08	2.61 ± 0.11	2.8 ± 0.13	5.58 ± 0.30	0.53 ± 0.05	1.21 ± 0.12	$F_{rich:3,524} = 119.1$; $F_{abun:3,524} = 125.4$

^a Small (<1 ha); medium (1–3 ha); large (3–6 ha).

^b Rich., richness ± SE, measured as average number of birds per point observation.

^c Abun., abundance ± SE, measured as number of individuals per point observation.

^d All are significant at the 0.001 level.

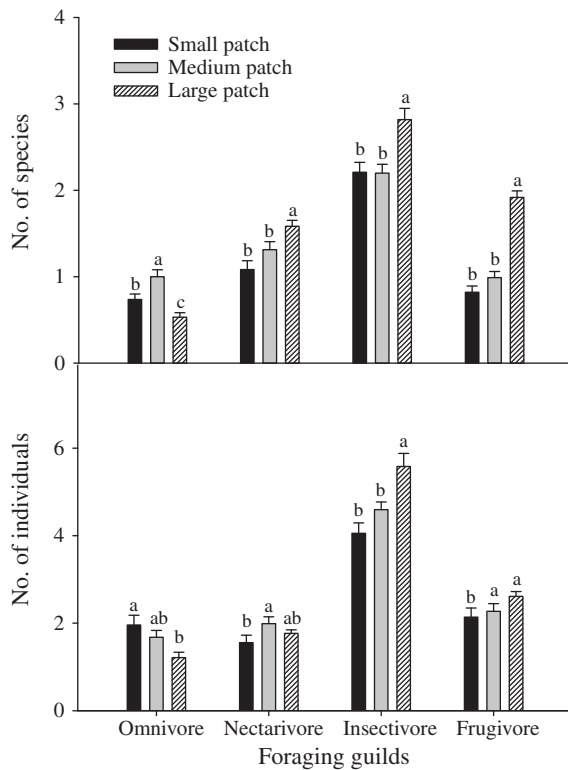


Fig. 6. Influence of patch area on richness (birds/point observation) and abundance (individuals/point observation) of birds classified by feeding guilds. Bars with the same letter are not significantly different at $p < 0.05$ (ANOVA). Comparisons are made within each category (i.e., Omnivore, Nectarivore, Insectivore, and Frugivore).

fragments in the rubber plantation matrix. The proportions of feeding guilds ($\chi^2 = 9.65$, $df = 3$, $p = 0.02$) and body sizes ($\chi^2 = 13.57$, $df = 2$, $p = 0.001$) in nature reserves were different to those in patches. Nectarivorous birds and small birds were most tolerant

of fragmentation, as the proportion of these animals in patches increased compared to those found in the nature reserve (Fig. 8). Large frugivorous and omnivores species were most vulnerable to fragmentation. For example, once widely distributed large frugivorous birds such as hornbills (Great hornbill *Buceros bicornis* and Oriental pied hornbill *Anthracoceros albirostris*) and several species of ground omnivores in the Pheasant family (Silver Pheasant *Lophura nycthemera* and Common Pheasant *Phasianus colchicus*) are absent from the fragment habitat surveyed here. The few large species persisting in patches are low in abundance. These include the mountain imperial pigeon (*Columba badia*) and lesser coucal (*Centropus bengalensis*).

4. Discussion

4.1. Bird diversity and forest fragmentation

This study has shown that deforestation and the concomitant fragmentation of Xishuangbanna's lowland forests has severely reduced the forest bird community outside nature reserves, with only a small proportion of reserve-resident species (73 of 265) found in fragments embedded in rubber plantations. Similar results were obtained by Aratrakorn et al. (2006) in plantations in southern Thailand. Because the majority of fragments in the rubber plantation were very small and severely disturbed, we predicted that only a few widespread disturbance specialists could inhabit them. Fortunately, this hypothesis was not supported, as shown by the low similarity in species composition within and across different forest patches. Instead, only four of the 73 resident species were distributed across all fragments, suggesting that the fragments in total support a relatively diverse assemblage of species able to tolerate highly disturbed forest conditions. Moreover, 24 migrants and three species of conservation concern in China were also recorded in these fragments (Appendix 2). Yorke (1984) also reported that the rubber systems in Malaysia play a complementary role in bird conservation.

Table 2
Bird richness and abundance by body-length across different patch areas in a rubber plantation.

Patch area ^a	Body length (cm)						ANOVA ^d
	<20 cm (Small)		20–40 cm (Medium)		>40 cm (Large)		
	Rich. ^b	Abun. ^c	Rich.	Abun.	Rich.	Abun.	
Small	3.71 ± 0.02	7.50 ± 0.34	1.24 ± 0.12	1.99 ± 0.17	0.13 ± 0.05	0.28 ± 0.11	$F_{rich:2,213} = 237.0$; $F_{abun:2,213} = 270.6$
Medium	3.72 ± 0.12	7.68 ± 0.29	1.32 ± 0.09	1.95 ± 0.12	0.19 ± 0.04	0.27 ± 0.07	$F_{rich:2,285} = 425.4$; $F_{abun:2,285} = 428.6$
Large	4.63 ± 0.11	7.98 ± 0.33	1.84 ± 0.10	2.69 ± 0.12	0.22 ± 0.04	0.44 ± 0.09	$F_{rich:2,393} = 633.7$; $F_{abun:2,393} = 346.8$

^a Small (<1 ha); medium (1–3 ha); large (3–6 ha).

^b Rich., richness ± SE, measured as average number of birds per point observation.

^c Abun., abundance ± SE, measured as number of individuals per point observation.

^d All are significant at the 0.001 levels.

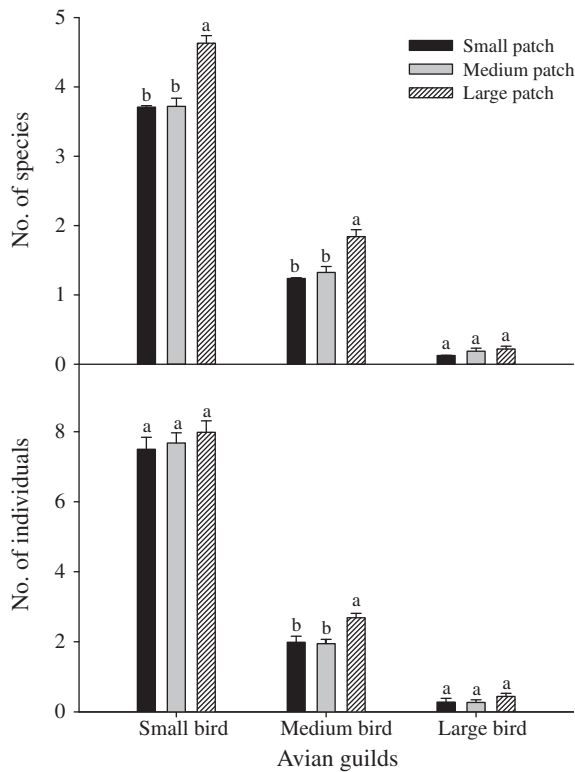


Fig. 7. Influence of patch area on richness (birds/point observation) and abundance (individuals/point observation) of bird species classified by body length. Different letters above each bar indicate statistical significance at $p < 0.05$ (ANOVA). Comparisons are made within each category (i.e., small bird, medium bird, and large bird).

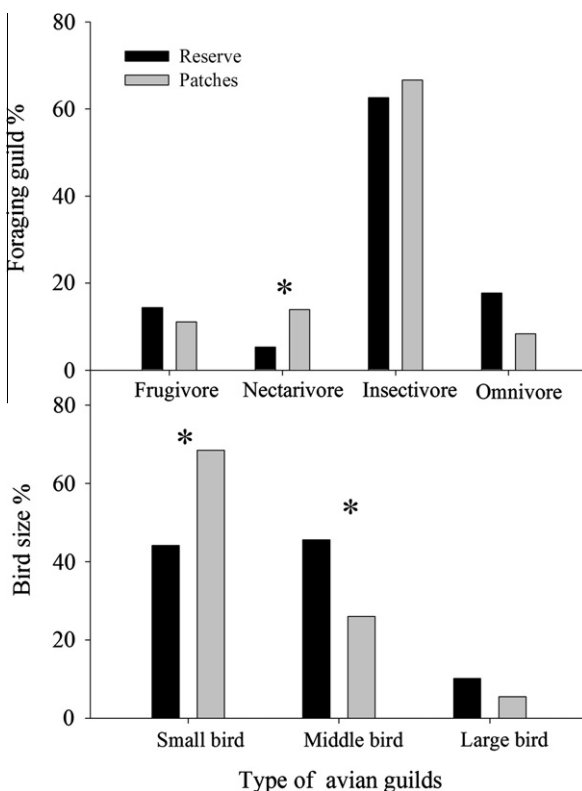


Fig. 8. Species composition of fragments and reserves. For each avian guild, the higher the proportion of species in the fragment, the higher the tolerance to fragmentation (*: significant at $p < 0.05$).

Our second prediction was that because not all bird species are equally vulnerable to disturbance (Aratrakorn et al., 2006; Gray et al., 2007), the similarity in species composition between the wet season, when rubber is tapped daily and also the decreased food resources by herbicide, and the dry season would be low. This was confirmed, and although we observed 73 resident bird species inhabiting these extremely small fragments, most used the patches only seasonally. We further analyzed the effect of season on avian communities by comparing species richness and abundance between the wet and dry seasons. We found both richness and abundance were significantly lower in the wet season (Fig. 5). Many of the species observed in the wet season occurred in the dry season, however, about 40% of the species in the dry season were not found in the wet season. Studies in other agroforestry systems have also detected significant shifts in bird abundance from dry to wet seasons (Calvo and Blake, 1998; Greenberg et al., 1997). As indicated by Marsden and Pilgrim (2003), apart from the influence of food sources, the birds' ability to persist over a longer time in fragments may also depend on the availability of nesting sites, as well as competition and predation, for which we do not have good assessments to date.

Insectivores were the most tolerant of disturbance and dominated all patches (Table 1, Fig. 6). The 20 species unique to large patches (3–6 ha) were mostly ground-dwelling insectivores (Appendix 2). As shown by Pattanavibool and Dearden (2002), and Stouffer and Bierregaard (1995), ground-dwelling insectivores are active in the interior zone of forest patches and this prohibits their use of smaller patches. It is therefore unsurprising that they were missing from the small patches sampled here (<3 ha). Because they feed on the ground, these species are also more sensitive to disturbance (Cockle et al., 2005), and this probably explains why 20 such species were only found in larger patches (3–6 ha) in which the anthropogenic effects are buffered.

The review of Scales and Marsden (2008) showed that intensive management and a decrease of stratum richness would decrease biodiversity in agroforestry systems. In contrast to the evidence that insectivorous bird populations use structurally more complex jungle-rubber or coffee plantations (Beukema et al., 2007; Greenberg et al., 1997), there were almost no insectivores using the monoculture plantations in our study site (X. Chang, pers. observ.). In our study area, native woody plants are removed and even herbaceous species are controlled by frequent use of herbicides. These structurally simple plantations prevent bird exploitation of these systems, especially the forest floor, and understory bird species are more vulnerable to plantation ecosystems (Cockle et al., 2005; Faria et al., 2006). Furthermore, the frequent use of pesticides in rubber plantations in our study site has probably removed many of the food items (such as arthropods and caterpillars) for insectivores.

Smaller birds were most tolerant of fragmentation (Table 2). Sodhi et al. (2004b) also found that larger-bodied species, especially large frugivores such as hornbills, are the most susceptible to fragmentation (Table 2, Fig. 7). Because fruit availability is variable in time and space, large frugivorous birds require larger and more continuous forest environments to facilitate seasonal foraging. Unlike oil palm, which can provide fruit for large frugivores and partridges (Sheldon et al., 2010), rubber plantations produce no fleshy fruits. Rubber plantations in our study site are also intensively managed to maximize yield and the frequent presence of humans may discourage wary species. These factors may explain why some once very common large frugivorous birds, such as the hornbill (*B. bicornis*), have gone locally extinct in the fragments surveyed. Other studies have also suggested that forest degradation and fragmentation result in the elimination of large frugivorous birds from parts of tropical Asia (Corlett, 2007; Waltert et al., 2004). A critical requirement therefore is that some large frag-

ments of natural forest habitat must remain protected in order to provide a refuge for frugivorous birds.

Fifteen fragments is only a tiny fraction of all the forest fragments in the whole Xishuangbanna region, so these results must be treated with caution. The locations of the sample fragments relative to other land use types and especially to large tracts of original forest may have a large influence on results, as they might have been sources for bird recorded in the rubber matrix (Beukema et al., 2007; Cockle et al., 2005). Besides, differences in topography were also reported to influence bird assemblages in agroforests (Thiollay, 1995).

In conclusion, our study shows that even extremely small fragments can play an important role in the conservation of bird diversity. However, the patches included in this study were too small to support an annual stable population; especially for the frugivorous and large bird species which are the most vulnerable to fragmentation. We suggest that future studies should examine the minimum habitat requirements for sensitive species across the region, looking not only at the sizes of individual forest patches but also the proportion of forest in the landscape as a whole.

4.2. Conservation implications

The tropical forest landscapes that once dominated the lowland regions of Xishuangbanna have now mostly been replaced by agricultural ecosystems. Empirical studies into the effects of land-use change on biodiversity are lacking from this region and birds are an ideal taxonomic group for such investigations. Our study revealed the encouraging finding that 97 avian species remain in these highly disturbed fragments. Similar to reports elsewhere (Harvey et al., 2006; Sekercioglu et al., 2007), forest fragments within rubber plantation probably provide complementary habitat and resources for a significant portion of the native bird community. However, as indicated by the very low similarity index in species composition, the current area of remnant habitat is not large enough to support an independent and sustainable avian community, especially one comprising large frugivores and forest-restricted species (Maas et al., 2009). To conserve the remaining avian diversity conservation policies should prevent further fragmentation of habitat and improve patch quality by connecting larger patches or by decreasing patch disturbance to facilitate the movement of birds between patches.

Although plantations have proved to be able to contribute to biodiversity conservation, the occurrence of native fauna in many agroforestry systems, such as oil palm and rubber plantations, is presumed to be associated with structural complexity within the plantation (Beukema et al., 2007; Najera and Simonetti, 2010; Sheldon et al., 2010). To establish a 'biodiversity friendly rubber system', we suggest future efforts across the Xishuangbanna region should focus on decreasing rubber plantation management activities (e.g., ceasing herbicide use) to facilitate the reestablishment of some woody and herbaceous plants in rubber plantations. We also suggest that larger forest patches be conserved to specifically meet the needs of larger and fragmentation-sensitive frugivores. However, the exact minimum habitat requirements of these larger frugivorous species in Xishuangbanna remain unknown and this will be the subject of our future work in this area.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.biocon.2012.09.024>.

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