Conservation value of human-modified forests for birds in mountainous regions of south-west China

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Summary

Mountain systems are often recognised as biodiversity hotspots; however, most montane forests are human-modified. Understanding the conservation value of human-modified forests is essential to mountain biodiversity conservation. In this study, mist-nets and point-counts were used to compare the bird communities of primary forests on ridges in the Ailao Mountains of Yunnan, China, and secondary forests, firewood forests, and pine plantations in nearby lower elevation zones. We found that community turnover among habitats was very high using both methods. Although the percentage of forest interior species and insectivores in each habitat was higher in the primary forests than in the human-modified forests, relatively high percentages of forest interior species and insectivores were recorded in the human-modified forests. Moreover, many bird species, forest interior species, insectivores and babblers were only recorded in the human-modified forests. Our study indicated that the primary forests are important for sustaining mountain bird diversity. However, given that primary forests are restricted to ridges, secondary forests, firewood forests, and pine plantations at lower elevations also play important roles in bird conservation in mountainous regions by increasing landscape diversity and partially offsetting species loss from primary forests. Therefore, conservation efforts should also be concentrated on human-modified forests at lower elevations in mountainous regions in south-west China.

Introduction

Mountain systems are often considered biodiversity hotspots because they contain high species diversity and a highly disproportionate number of endemic and threatened species (Lomolino 2001, Blyth *et al.* 2002, Körner and Spehn 2002, Sekercioglu *et al.* 2012). Mountains cover approximately 27 % of the Earth's land surface (Blyth *et al.* 2002) and host half of its 34 biodiversity hotspots (Myers *et al.* 2000, Mittermeier *et al.* 2005, Spehn and Körner 2005). At the same time, montane forests are considered among the most threatened global ecosystems (Aldrich *et al.* 1997) and the most fragile ecosystems in Asia (McNeely *et al.* 2009). Montane primary forests are gradually decreasing in area because of the overuse of natural resources, inappropriate infrastructural development, deforestation, and the impacts of natural hazards and massive afforestation (Lan and Dunbar 2000, Blyth *et al.* 2002, Brockerhoff *et al.* 2008). Human-modified landscapes play important roles in biodiversity conservation (Gardner *et al.* 2009, 2010, Peres *et al.* 2010, Melo *et al.* 2013). Mountainous regions are always dominated by human-modified forests, usually including secondary forests, firewood forests, and plantations. Thus, understanding the impacts of human-modified forests on mountain biodiversity is critical to conservation.

China is one of the world's "megabiodiversity countries" (McNeely *et al.* 1990). Mountains cover 46% of China's land surface (Jiang 2008), where they are conservation priority areas because they are critical for protecting biodiversity (Tang *et al.* 2006). By the end of 2004, a total of 2,194 national and local nature reserves were established in China (Ministry of Environmental Protection of the People's Republic of China 2005). Of those nature reserves, 73.8% are located in mountainous regions, which cover 83.0% of all protected areas (1.5 million km²) (Tang *et al.* 2006). In south-west China, most of the mountain nature reserves have concentrated their conservation efforts on forests in higher elevation zones, where primary (old growth) forests persist. In lower elevation zones, primary forests have been replaced by secondary forests, firewood forests, plantations, and open areas as a result of long–term human disturbance. However, little information exists about the conservation value of these human-modified montane forests in south-west China.

Primary forests are crucial to biodiversity conservation (Gibson *et al.* 2011). The conservation value of forests is usually assessed based on the number (or proportion) of species that occupy mainly primary forests (Chazdon *et al.* 2009, Dent and Wright 2009, Barlow *et al.* 2010). However, this method of assessing the conservation value of human-modified forests is difficult in mountainous regions of China because the primary forests used for comparison are unavailable at most elevation zones, except on mountain ridges. This challenge may explain why so few relevant studies have been conducted in the mountainous regions of China. Furthermore, although there are primary forests for comparison with the degraded and plantation forests in several elevation zones (e.g. Soh *et al.* 2006), these results cannot be simply extrapolated to human-modified forests at other elevations because of rapid species turnover along montane elevation gradients (Blake and Loiselle 2000, Kattan *et al.* 2004, 2006, Jankowski *et al.* 2009, Wu *et al.* 2010). Non-primary forests may be indispensable for some primary forest species occurring only in these elevation zones, although these forests may harbour a relatively low proportion of primary forest species. Thus, the conservation value of human-modified forests may be underestimated if they are only valued on the basis of the number (or proportion) of primary forest species.

We hypothesise that the value of human-modified forests for conservation of biodiversity is underestimated in mountainous regions. In this study, we examine this claim using two ways of evaluating the value of secondary forests, firewood forests, and pine plantations for montane avifauna in the Ailao Mountains of Yunnan, China. First, we examined the degree of dissimilarity in species composition between the human-modified and primary forests. Second, we considered how many species and exclusive species were present in the human-modified forests. A high dissimilarity in species composition between these two forest types would mean that human-modified forests are also important for regional species-diversity conservation, particularly when the species richness of the human-modified forests is high. We used birds as our focal taxon because they are good environmental indicators and are relatively easy to sample (Bibby *et al.* 2000, Carignan and Villard 2002). The human disturbance patterns in south-west China are strongly evident in the Ailao Mountains, where primary forests are present only on mountain ridges, and human-modified forests at lower elevations are threatened by deforestation or further degradation and are even being converted to cropland. Our study will provide information on conserving birds in the Ailao Mountains and will allow managers, planners, and policy makers to design more effective strategies for bird conservation in mountainous regions.

Material and methods

Study sites

The study was conducted in the Ailao Mountains situated in central Yunnan Province, China (24°18′–24°29′N, 101°08′–101°19′E) (Figure 1). This mountain range is part of the Indo-Burma biodiversity hotspot and is recognised as a high-priority area for biodiversity conservation by Conservation International, BirdLife International, and WWF (Olson and Dinerstein 1998, Myers
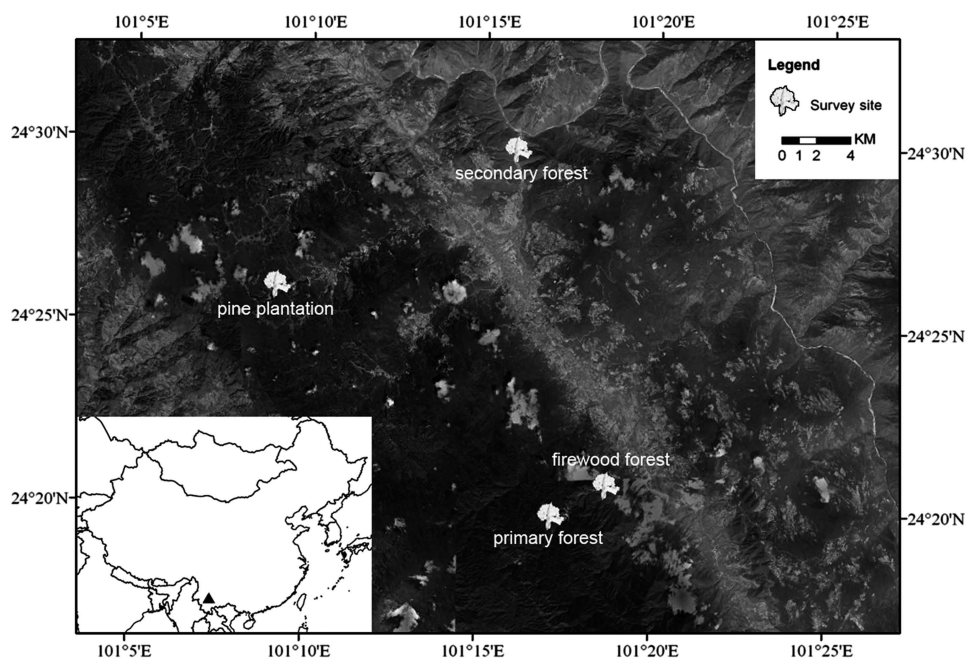
Figure 1. Location of study sites in the Ailao Mountains, Yunnan Province, China, darker areas indicate high forest cover (revised from Google Earth).

et al. 2000, Mittermeier *et al.* 2005, Chan *et al.* 2009). A national nature reserve exists at higher elevations in the mountain range.

The Ailao Mountains form the border between the two main physiographic regions of Yunnan: to the west, the Western Valley land is a region of deep valleys and high mountains, and to the east, the Yunnan–Guizhou plateau consists of hills atop a high plateau (average elevation c.2,000 m) (Wang *et al.* 2000). The general climate of the Ailao Mountains is subtropical montane with warm, humid conditions at the base (Wang *et al.* 1988). On the eastern slope of the Ailao Mountains, annual precipitation reaches a maximum of 1,700 mm at the top and a minimum of 700 mm in the river valley, with over 85% of the precipitation occurring between May and October (Wang *et al.* 1988). The top of the Ailao Mountains has maximum and minimum mean monthly temperatures of 10–17°C and 0–6°C, respectively, with the mean annual temperature below 12°C. Five months have a mean monthly temperature exceeding 22°C, and none have a mean monthly temperature below 10°C in the river valley (Wang *et al.* 1988). The Ailao Mountains contain the largest and most continuous subtropical evergreen broad-leaved forests (over 45,000 ha) in China (Pang *et al.* 1988).

Our study sites were located in four forest types: primary forests on the ridges as well as pine plantations, firewood forests, and secondary forests on the eastern slope. These sites were distributed along an elevation range of 750 m to 2,800 m (Figure 1). Primary forests (2,400–2,800 m elevation) mainly consisted of native evergreen broad-leaved trees, such as *Castanopsis remotidenticulata*, *Lithocarpus jingdongensis*, *L. xylocarpus*, *L. pachyphlloides*, *Schima argentea*, *Manglietia insignis*, *Tetracentran sinense*, *Machilus viridis* and *M. longipedicellata*. Tree species of Fagaceae, Theaceae, Lauraceae and Magnoliaceae were dominant in the primary forest. These forests were in the core area of the Ailaoshan National Nature Reserve.

Pine plantations (2,350–2,550 m) were located on the edge of the ridges and mainly comprised native Armand's pine *Pinus armandii* 11–16 years old in the experimental zone of the Ailaoshan

National Nature Reserve. Most of the pine plantation area was used for shifting cultivation. Shifting cultivation was forbidden when the Ailaoshan Nature Reserve was established in 1988, and Armand's pine was planted for harvesting pine nuts between 1990 and 1995. Grazing, but not tree harvesting, was permitted in the pine plantations.

Firewood forests (1,850–2,400 m) were mainly secondary evergreen broad-leaved forests comprised of Nepalese alder *Alnus nepalensis* forest. Common tree species of the firewood forests are *Castanopsis delavayi, C. orthacantha, Cyclobalanopsis glaucoides, ithocarpus. dealbatus, L. leucostachyus, L. polystachyus, Phoebe macrocarpa, P. puwenensis, Cinnamomum glanduliferum, Lindera latifolia, Schima argentea* and *Ternstroemia gymnanthera*. Annually, some larger trees were logged for firewood by local residents, and these forests were also the main area for local cattle grazing. Firewood forests have undergone serious degradation in the past 30 years.

The secondary forests (750–1,300 m) were mainly arid woodlands, consisting of secondary evergreen broad-leaved trees, such as *C. kerrii* and *Buchanania latifolia*. The secondary forests were part of the Konglonghe Municipal Nature Reserve (China has nature reserves at national, provincial, municipal, and county levels), which was established in 2003. Firewood cutting and grazing are forbidden in secondary forests but occurred occasionally.

Bird sampling

We sampled the bird community using point-counts and mist-nets. The combination of both methods has been considered the most effective for monitoring bird assemblages (Rappole *et al.* 1998, Blake and Loiselle 2001, Derlindati and Caziani 2005). Point-counts are a powerful method for measuring bird species richness, relative abundance, and density (Verner and Ritter 1985, Buckland *et al.* 2001, Gale *et al.* 2009) and are particularly suitable for counting birds in spatially complex habitats (Bibby *et al.* 2000). They have been widely used in bird surveys in subtropical forests (Whitman *et al.* 1997).

Unlimited radius point-counts were performed during the peak period of bird activity between sunrise and 3.5 h after sunrise. Sampling occurred between 7 March and 8 May (late dry season, the breeding season), and 3 November and 10 December (dry season, the nonbreeding season) of 2006. Each count was 10 min in duration. During this period, all birds seen and heard were recorded, and the horizontal distance from the observer to each detected bird was estimated. Over-flying birds and uncertain identifications were excluded from the analysis. Because of the extremely rugged terrain and limited access, points were established on pre-existing trails (> 95%) or low–traffic roads (Naidoo 2004, Simons *et al.* 2006, O'Dea and Whittaker 2007). We recorded the time and location of each point using a Summit GPS receiver (Garmin, Olathe, KS, USA). Points were located at least 200 m apart and at a minimum of 100 m from the edge of each habitat type to reduce edge effects. Point-counts were only performed on days with little or no wind, rain, or fog. A total of 168 points (42 per habitat type) were surveyed. Each survey point was visited twice, once each during the spring and winter sampling periods. Before commencing fieldwork, we spent four months becoming familiar with visual and auditory identification of local birds and practising distance estimation during a 3-week pilot study in 2005.

We also used mist-nets to survey the understorey bird species. In the same periods during which the point-counts were performed, we erected mist-nets at two sites at least 1 km apart in each habitat type. Ten mist nets (2.5 × 12.0 m, 36-mm mesh) were placed at each site at least 40 m apart. At any given site, the nets were opened from mid-afternoon to dusk on day 1, dawn to dusk on days 2 and 3, and dawn to late morning on day 4. During the rains, the nets were closed to prevent bird mortality. We recorded the time at which each net was opened and closed. Total net hours in each habitat were calculated as the total hours of 10 mist-nets opened in two sites of each habitat. Total net hours in the secondary forests, firewood forests, pine plantations, and primary forests were 753, 745, 771, and 760 in the breeding season and 653, 656, 647, and 673 in the

non-breeding season, respectively. All mist-netted birds were identified, weighed, banded, and released. Recaptured individuals were excluded from the total counts. Bird taxonomy followed Yang *et al.* (1995) and Yang and Yang (2004).

Data analysis

Data for all birds detected in our study region were analysed, with the exception of nocturnal birds, aquatic birds, and raptors. To limit systematic bias among habitats because of the sensitivity of detection to particular combinations of sampling method and forest type, we treated mist-net and point-count data separately (Barlow *et al.* 2007). Similar results from these two very different methods indicate strong support for our conclusions. Because the similarity matrices of community structure among habitat types from the breeding and nonbreeding seasons were highly correlated (Mantel test: point-counts, $r = 0.81$, $P = 0.002$; mist-nets, $r = 0.92$, $P = 0.001$), we combined data for the two seasons. Data on forest interior species and insectivores were extracted from Yang *et al.* (1995), MacKinnon and Phillipps (2000), Robson (2000), and Yang and Yang (2004). Forest interior species were those reported to mainly occupy interior forests.

For mist-net and point-count data, we constructed sample-based rarefaction curves rescaled to the number of individuals to compare species richness among the habitats (Colwell *et al.* 2004). Although rarefaction allows interpolation to compare species richness at smaller sample sizes, it cannot be used for extrapolation. Hence, it does not provide an estimate of asymptotic species richness (Gotelli and Colwell 2001). However, based on the distribution of rare bird species in the assemblages, non-parametric species estimators were used to estimate "true" species richness for each habitat (Colwell and Coddington 1994). We used EstimateS v. 8.0 (Colwell 2009) to calculate the means of the four commonly used abundance-based estimators (ACE, CHAO1, JACK1, and bootstrap).

We used additive diversity partitioning to calculate species turnover (i.e. β diversity) among four habitats ($\beta_{habitat}$) based on a null model. We used species richness in our study to account for the effects of species richness, when sample weights were equal (Jost *et al.* 2010). We used PARTITION and selected the individual-based randomisation routine (Crist *et al.* 2003). The randomisation process was repeated 1,000 times to form a null distribution of each scale-specific alpha and beta value. The statistical significance of each diversity component was assessed by the proportion of null values greater than (or less than) the observed value (Crist *et al.* 2003, Gering *et al.* 2003).

The multi-response permutation procedure (MRPP) was used to determine the degree of similarity among communities represented by each habitat for both methods (McCune and Grace 2002). MRPP assesses whether species composition within groups is more similar than expected by chance (McCune and Grace 2002). It yields a probability of significance and a measure of effect size, *A* (the chance-corrected, within-group agreement). It is preferred to similar techniques because it avoids distributional assumptions. Similarity matrices from mist-nets and point-counts were compared using the Mantel test and Mantel's asymptotic approximation method (McCune and Grace 2002). The square root transformation was used for abundance data to reduce the influence of the most dominant species in all community structure analyses, and the Sørenson (Bray-Curtis) method was chosen as the distance measure for MRPP and the Mantel test. All rarefaction analysis and species estimator calculations were performed using EstimateS. MRPP and the Mantel test were carried out using PC-ORD v. 4.14 (McCune and Mefford 1999).

Results

We recorded 3,055 individuals of 150 species from point-counts and captured 1,745 individuals of 128 species. Mist-nets added 30 species to the total. The similarity matrices of community structure among habitat types from point-counts and mist-nets were highly correlated (Mantel test: $r = 0.87$, $P = 0.001$).

Species richness

For both methods, rarefaction curves showed that species richness was the highest in the firewood forests and lowest in the pine plantations (Figure 2). Average species richness estimates using point-counts gave similar results, although mist-net data did not differ significantly among the four habitat types (Table 1). The results of additive diversity partitioning showed that $\beta_{habitat}$ were significantly higher ($P = 0.00$) than expected by chance and accounted for 54.87% and 57.44% of the total species diversity in point-counts and mist-net, respectively (Table 2). The point-count and mist-net methods showed that 49% and 55%, respectively, of bird species were unique to one of the four habitats. Exclusive species were most prevalent in the secondary forests and least in the pine plantations using both methods. Each habitat contained an average of 45% of all recorded species using point-count data, with a maximum of 49% in firewood forests to a minimum of 40% in the pine plantations. Using mist-net data, each habitat contained an average of 41% of all captured species, ranging from 47% in the firewood forests to 37% in the secondary forests (Table 1, Figure 3).

Forest interior species and insectivores

In total, we recorded 51 forest interior species (906 individuals) and 90 insectivore species (1,477 individuals) from point-counts, and 45 forest interior species (560 individuals) and 78 insectivore species (1,239 individuals) were captured (Appendix S1 in the online supplementary material). The results of additive diversity partitioning showed that $\beta_{habitat}$ were significantly higher
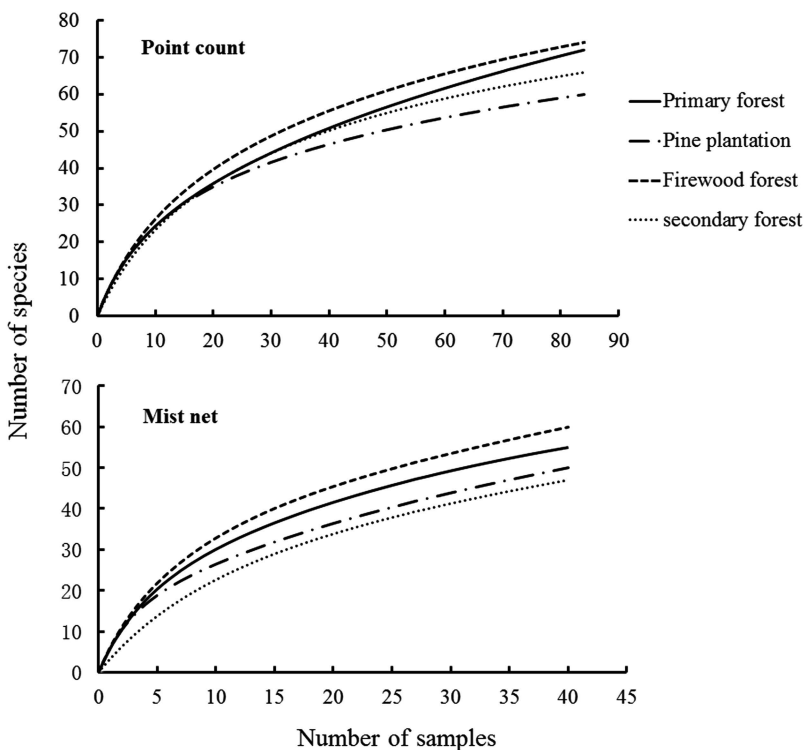

Figure 2. Sample-based rarefaction curves for point-count and mist net data for four montane forest habitats. The x-axis indicates the number of samples.

Table 1. Nonparametric species richness estimates and number of exclusive species in each of the four habitats from point-count and mist net data.

		Primary forests	Pine plantations	Firewood forests	Secondary forests
Point-counts					
All species	Ind_{obs}	841	852	653	709
	Sp_{obs}	72	60	74	66
	Sp_{est}	91.21 ± 4.54^{ab}	68.46 ± 3.71^{c}	90.24 ± 2.90^{a}	77.85 ± 3.06^{bc}
	Sp_{exc}	18	8	15	32
Forest interior	Ind_{obs}	307	212	173	332
	Sp_{obs}	31	15	20	21
	Sp_{est}	35.12 ± 1.74^{a}	18.33 ± 0.61^{c}	24.00 ± 0.58^{b}	26.71 ± 1.42^{abc}
	Sp_{exc}	13	1	4	11
Insectivores	Ind_{obs}	433	402	350	292
	Sp_{obs}	51	32	41	41
	Sp_{est}	66.01 ± 3.61^{a}	37.35 ± 2.33^{c}	51.97 ± 1.67^{abc}	48.60 ± 2.16^{b}
	Sp_{exc}	14	3	10	18
Mist nets					
All species	Ind_{cap}	520	551	426	248
	Sp_{cap}	55	50	60	47
	Sp_{est}	69.57 ± 2.33^{a}	78.60 ± 8.03^{a}	88.71 ± 11.00^{a}	76.80 ± 12.68^{a}
	Sp_{exc}	16	12	18	25
Forest interior	Ind_{cap}	264	135	121	40
	Sp_{cap}	27	18	14	15
	Sp_{est}	33.20 ± 1.11^{a}	28.34 ± 3.05^{a}	21.70 ± 2.61^{a}	25.63 ± 4.73^{a}
	Sp_{exc}	11	6	1	8
Insectivores	Ind_{cap}	418	467	275	79
	Sp_{cap}	37	28	38	29
	Sp_{est}	44.82 ± 1.47^{a}	38.36 ± 2.73^{a}	64.79 ± 12.60^{a}	52.42 ± 7.64^{a}
	Sp_{exc}	10	3	11	16

Sp_{obs} and Ind_{obs} represent the total number of species and individuals observed, and Sp_{cap} and Ind_{cap} represent the total number of species and individuals captured, respectively. Sp_{est} represents the average species richness estimate ($\pm$SE) (ACE, CHAO1, Jack1, and Bootstrap, see Colwell and Coddington, 1994). Sp_{exc} represents exclusive species. Average species richness estimate of habitat types are ranked (a–c); significance of differences among the habitats was tested by the paired *t*-test corrected by the Benjamini and Hochberg method (1995), where values did not differ significantly at $P = 0.05$, their ranks are the same.

($P = 0.00$) than expected by chance for both forest interior species and insectivore species. For forest interior species, $\beta_{habitat}$ accounted for 55.3% and 52.98% of the total species diversity in point-counts and mist-net respectively. For insectivore species, $\beta_{habitat}$ accounted for 54.12% and 57.28% of the total species diversity in point-counts and mist-net, respectively (Table 2). For the point-count and mist-nest methods, at least half of the forest interior species (29 and 26, respectively) and insectivore species (45 and 40, respectively) were unique to one of the four habitats. For both methods, exclusive forest interior species in each habitat were recorded most often in the primary forests, while exclusive insectivore species in each habitat were recorded most often in secondary forests (Table 1).

Average species richness estimates for the forest interior species and insectivores using point-counts were highest in the primary forests and lowest in the pine plantations, while they were not significantly different among the four habitat types using mist-nets (Table 1). More than half of the forest interior species (point-counts, 61%; mist-nets, 60%) were recorded in the primary forests. The average number of forest interior species found in each of the other habitats was 37% (29–41%) using point-counts and 35% (31–40%) using mist-nets (Figure 3). For insectivore species in each of the habitats, an average of 46% (36–57%) was found using point-counts and 42% (36–49%) using mist-nets (Figure 3).

Table 2. Additive partitioning of species diversity (expressed as species richness) of the bird assemblage in four habitats.

		Total			Forest interior			Insectivores		
		observed	%	P^*	observed	%	P	observed	%	P
Point										
α_{point}	Within points	4.89	3.26	1	2.47	4.84	1	3.6	4	1
β_{point}	Among points	62.8	41.87	1	20.33	39.86	1	37.69	41.88	1
$\beta_{habitats}$	Among habitats	82.31	54.87	0	28.2	55.3	0	48.71	54.12	0
γ	Total	150			51			90		
Net										
α_{point}	Within habitat	7.97	06.23	1	3.83	8.51	1	6.22	7.98	1
β_{point}	Among points	46.5	36.33	1	17.33	38.51	1	27.1	34.74	1
$\beta_{habitats}$	Among habitats	73.53	57.44	0	23.84	52.98	0	44.68	57.28	0
γ	Total	128			45			78		

$^*P = 0.00$ indicates that the observed diversity value is significantly larger than the randomized datasets produced ($P = 0.00$). $P = 1.00$ indicates that diversity value of the randomized datasets produced is significantly larger than that observed ($P = 0.00$).

For the point-count and mist-net data, the highest percentage of forest interior and insectivore species was in the primary forests (Figure 4). With point-counts, a similar percentage of forest interior species (25–27%) was recorded in the secondary forests, firewood forests, and pine plantations, while the mist-net data found a higher percentage of forest interior species in the pine
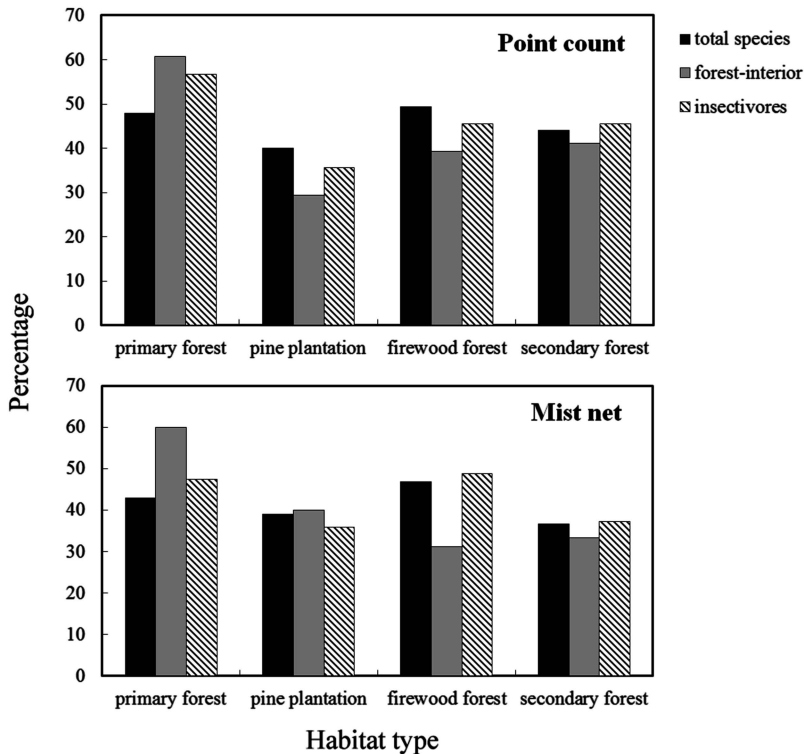

Figure 3. Representation of all bird species, forest interior species, and insectivores in each of the four habitat types as a percentage of all birds in all habitats in the Ailao Mountains, China.
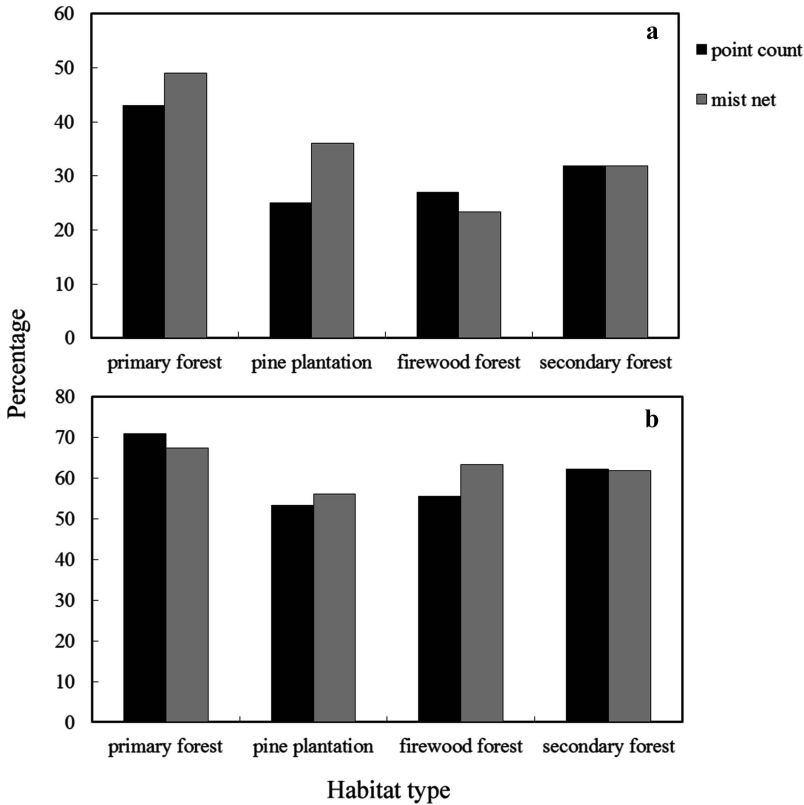
Figure 4. Percentage of forest interior species (a) and insectivore species (b) in each of the four montane habitat types in the Ailao Mountains, China.

plantations and secondary forests than in the firewood forests (Figure 4a). Similar percentages of insectivore species (53–62%) were recorded in the pine plantations, firewood forests, and secondary forests using both methods (Figure 4b).

Babbler avifauna

We recorded a total of 44 species of babbler combining records of the two methods. Eighteen of these species were recorded only in the human-modified forests. The number of babbler species was highest in the primary forests and lowest in the secondary forests (Figure 5). The number of exclusive babbler species was higher in the primary forests and secondary forests than in the pine plantations and firewood forests.

Community similarities among habitats

MRPP indicated a significant difference in species composition among all habitat types using both point-count and mist-net data ($P < 0.001$ for all pairwise compositions; Table 3). For both methods, bird community composition differed the most between primary and secondary forests and the least between pine plantations and firewood forests (Table 3). Furthermore, correlation analysis indicated a strong correlation between elevation distance between habitat types and the community dissimilarity using both mist-net (Rho = 0.991, $P = 0.000$) and point-count (Rho = 0.951, $P = 0.004$) data,
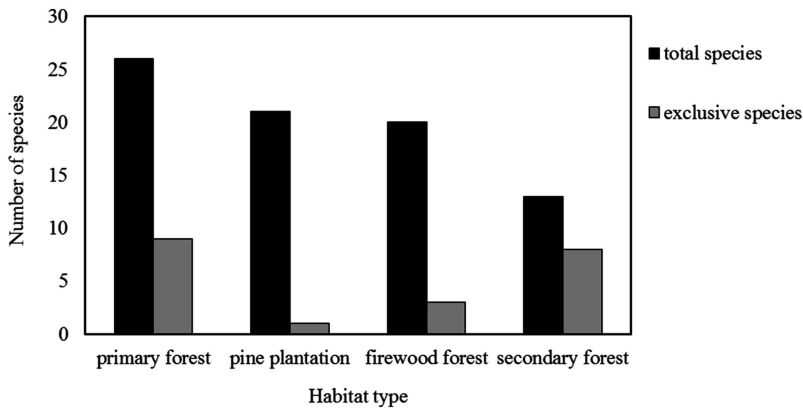
Figure 5. Number of all and exclusive babbler species recorded by point-counts and mist nets in each of the four habitat types in the Ailao Mountains, China.

while there was no relationship between geographic distance between sites and community dissimilarity using either mist-net (Rho = −0.101, $P = 0.849$) or point-count (Rho = −0.057, $P = 0.914$) data.

Discussion

Primary forests are crucial to biodiversity conservation (Gibson *et al.* 2011). Our study indicated that the primary forests were very important for sustaining mountain bird diversity. However, given that the primary forests could only be found on ridges in the Ailao Mountains, the results of our study revealed that secondary forests, firewood forests, and pine plantations at lower elevations also played important roles in bird conservation in mountainous regions by increasing landscape diversity and partially offsetting species loss from primary forests.

Maintaining regional bird diversity

Our results indicated that all four habitats in our study were important to maintaining regional bird diversity. Species turnover is rapid along elevation gradients in mountainous regions

Table 3. Sørensen distances and multi-response permutation procedure results with chance-corrected within-group agreement statistics (A) for point-count data (above the diagonal) and mist net data (below).

		Primary forests	Pine plantations	Firewood forests	Secondary forests
	Elevation (m)[a]	2657	2444	2085	939
			0.614	0.683	0.902
Primary forests	2699	-	0.123	0.124	0.169
		0.574		0.555	0.886
Pine plantations	2444	0.116	-	0.068	0.144
		0.710	0.605		0.825
Firewood forests	2020	0.159	0.114	-	0.106
		0.927	0.905	0.804	
Secondary forests	904	0.185	0.200	0.187	-

[a]Average elevation of all locations of point-counts or mist nets in each habitat.
For all pairwise comparisons, results indicated a significant difference between habitats ($p < 0.001$). A describes within-group homogeneity compared with random expectation (McCune and Grace, 2002). For A, when all items are identical within groups, then $A = 1$, the highest possible value; if heterogeneity within groups equals expectation by chance, then $A = 0$; and if there is less agreement within groups than expected by chance, then $A < 0$ (McCune and Grace 2002).

(Patterson *et al.* 1998, Blake and Loiselle 2000, Kattan *et al.* 2004, 2006, Jankowski *et al.* 2009, Wu *et al.* 2010). As expected, we found that species composition differed significantly among the habitat types at different elevations (Table 3), and species turnover among four habitats (i.e. $\beta_{habitat}$) accounted for most of the total species diversity (Table 2). Each habitat only represented some of the recorded species (averages of 45% for point-counts and 41% for mist-nets) (Table 1, Figure 4). Bird species richness in the secondary and firewood forests were higher or, at least, not lower than that in the primary forests (Table 1, Figure 2). Most of the recorded species were unique to one of the four habitats. Even in the pine plantations with the fewest exclusive bird species, there were eight exclusive species using point-counts and 12 using mist-nets (Table 1). Each habitat in our study was indispensable to maintaining regional bird diversity. Similar results were also found in Hainan Island, China (Zou *et al.* 2012). Our results showed that bird community dissimilarity to primary forests on the ridges of the Ailao Mountains increased rapidly with elevation distance (Table 3). Thus, to maintain regional bird diversity, secondary forests in the submontane region had higher conservation value. This finding was not consistent with other studies in which all habitat types were at similar elevations and that indicated that non-primary forests near primary forests had higher conservation value (Peh *et al.* 2006, Chazdon *et al.* 2009, Dent and Wright 2009).

Offsetting species loss from primary forests

The conservation value of human-modified forests is usually assessed based on the number (or proportion) of primary forest species they contain (Chazdon *et al.* 2009, Dent and Wright 2009, Barlow *et al.* 2010). However, as mentioned above, primary forests only exist on ridges in the Ailao Mountains and most other mountains in south-west China. Thus, in our study, we do not know how many primary forest bird species are present in lower-elevation human-modified forests. Forest interior species and insectivores are known to be vulnerable to habitat modification and to be present in large proportions in primary forests (Canaday 1996, Sodhi *et al.* 2004, 2005, Waltert *et al.* 2005, Gray *et al.* 2007, Sekercioglu 2012). The percentage of forest interior species and insectivores can reflect the value of a habitat for bird conservation. The more forest interior species and insectivores in human-modified forests, the higher is the conservation value of these forests.

In agreement with these findings, our results showed that the highest percentage of forest interior species and insectivores were detected in primary forests (Figure 3). Thus, primary forests are clearly important for persistence of bird species diversity in mountainous regions. Some bird species recorded between 1976 and 1977 at lower elevations in the Ailao Mountains (Wei *et al.* 1988), such as Scaly-breasted Partridge *Arborophila chloropus*, Limestone Wren Babbler *Napothera crispifrons*, Streaked Wren Babbler *N. brevicaudata* and Brown-cheeked Fulvetta *Alcippe poioicephala*, were not recorded after 2004 (Wu *et al.* 2015). All of these bird species are usually restricted to primary forests. The loss of primary forests at lower elevations might be the reason that they disappeared from the region. Our results also indicated that pine plantations, firewood forests, and secondary forests at lower elevations partially offset species loss from primary forests. In our study, although the primary forests had disappeared from lower elevations because of long-term human disturbance, these human-modified forests still had 23–36% of extant forest interior species and 53–62% of extant insectivore species (Figure 4). Given that turnover of forest interior and insectivore species was rapid among four habitats and more than 30% of extant forest interior and insectivore species in our study were only recorded in these human-modified forests (Tables 1, 2), they also have important value in partially offsetting vulnerable species loss from primary forests.

Babblers (Timaliidae) are considered very sensitive to disturbance (Yong 2009, Li *et al.* 2013). The babbler avifauna of south-west China is one of the most diverse in the world and 45.5% of the total individuals are made up of babblers. Considering that 26 out of a total of 44 babbler species were recorded in the primary forests, these forests are important for sustaining babbler diversity.

Our distribution results for babbler species in the four habitat types (Figure 5) also indicated that the human-modified forests were important in partially offsetting babbler loss from primary forests. A total of 35 babbler species were recorded in the human-modified forests, of which 18 species were found only in these forests: Spot-breasted Scimitar-babbler *Pomatorhinus erythrocnemis*, Streak-breasted Scimitar-babbler *P. ruficollis*, Grey-throated Babbler *Stachyris nigriceps*, Chinese Babax *Babax lanceolatus*, Hwamei *Garrulax canorus*, White-browed Laughingthrush *G. sannio*, Greater Necklaced Laughingthrush *G. pectoralis*, Lesser Necklaced Laughingthrush *G. monileger*, Moustached Laughingthrush *G. cineraceus*, Silver-eared Mesia *Leiothrix argentauris*, White-browed Shrike-babbler *Pteruthius flaviscapis*, Spectacled Barwing *Actinodura ramsayi*, Blue-winged Minla *Minla cyanouroptera*, Spectacled Fulvetta *Alcippe ruficapilla*, White-bellied Yuhina *Yuhina zantholeuca*, Striated Yuhina *Y. castaniceps*, Brown-winged Parrotbill *Paradoxornis brunneus*, and Spot-breasted Parrotbill *P. guttaticollis*. All of the human-modified forests were composed of native tree species and had suitable understorey habitats, which may be the reasons that 35 babbler species still existed in these forests (Li *et al.* 2013, Zou *et al.* 2014).

Our mist-net results showed that the percentage of forest interior species in firewood forests was lower than that in pine plantations and secondary forests, probably because of understorey disturbance due to human clearing and cattle grazing. Many studies have found that forest interior species were less resilient to habitat modification and human disturbance (Renjifo 1999, Greenberg *et al.* 2000, Barlow *et al.* 2002, Sekercioglu *et al.* 2002, O'Dea and Whittaker 2007).

Conclusion

Through analysis of species richness and composition in four forest types, our study showed that primary forests on the ridges of the Ailao Mountains are important for sustaining bird diversity. However, given that primary forests are restricted to ridges, our study revealed that secondary forests, firewood forests, and pine plantations in lower elevation zones also play important roles in bird diversity conservation in the Ailao Mountains. Such information is useful in guiding mountain conservation plans.

Mountainous regions are crucial for protecting biodiversity and are conservation priority areas in China (Tang *et al.* 2006) and Asia (McNeely *et al.* 2009). However, because most current conservation efforts concentrate on the preservation of primary forests, which only exist in higher elevation zones in mountainous regions, the conservation value of human-modified forests in lower elevation zones are ignored or underestimated. In this study, we examined bird diversity in primary forests on ridges in the core area of the Ailaoshan National Nature Reserve, pine plantations growing Armand's pine adjacent to ridges in the experimental zone of the Ailaoshan Reserve, secondary forests in the premontane part of the Konglonghe Municipal Nature Reserve, and mid-montane firewood forests not within a reserve.

Human-modified forests at lower elevations are threatened by deforestation or further degradation. Some such forests are being converted to cropland (e.g. walnut, tea, banana, rubber, sugar cane) (Lan and Dunbar 2000), which has less conservation value (Petit and Petit 2003). Of note, the pine plantations in our study mainly comprised a native species with a mixture of indigenous broad-leaved tree species and had a higher conservation value than plantations with a single tree species or those growing exotic tree species (Aratrakorn *et al.* 2006, Peh *et al.* 2006, Brockerhoff *et al.* 2008, Farwig *et al.* 2008, Zou *et al.* 2014). In addition, firewood is still the main fuel for residents in mountainous regions in south-west China. The combination of an increasing human population and decreasing forest area puts unprecedented pressure on the remnant human-modified forests in lower elevation zones, which face further degradation. Mountain bird communities are sensitive to forest quality (Soh *et al.* 2006), and thus, deforestation and further degradation of human-modified forests in lower elevation zones will threaten birds in mountainous regions of south-west China.

In summary, the primary forests are very important for sustaining mountain bird diversity, and human-modified forests in lower elevation zones are also important in sustaining bird

diversity in mountainous regions. Our results therefore suggest that, beside primary forests, conservation efforts should also be concentrated on human-modified forests at lower montane elevations in south-west China.

Supplementary Material

To view supplementary material for this article, please visit https://doi.org/10.1017/S0959270916000058

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