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Natural forest at landscape scale is most important for bird conservation in rubber plantation

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ABSTRACT

Rubber is one of the most rapidly expanding monocultures in the tropics, and has precipitated biodiversity and ecosystem function loss. Identifying measures to improve biodiversity outcomes in rubber-forest mosaics is critical for tropical fauna. We evaluated how avian diversity responded to plantation- and landscape-level environmental variables. The most parsimonious model at the plot scale contained inter-tree planting distance for rubber, plantation age, and inverse distance weight of forest as predictors. The most supported model at the landscape scale contained both distance to forest patches larger than 100 ha and natural forest area as predictors. Model predictions indicated that natural forest area had the largest contribution to bird richness at landscape levels; avian diversity was projected to more than double when natural forest area increased from 25% to 75%. Frugivores and insectivores exhibited the strongest response to gains in natural forest area. Our results indicated that plantation smallholders could achieve biodiversity gains by retaining older trees and planting rubber trees with larger gaps, but that the most critical intervention is retaining large natural forest patches.

1. Introduction

Globally, farmland is replacing natural forest due to human population growth, changing diets, and agricultural expansion (Defries et al., 2010; Tilman and Clark, 2014). From 2001 to 2050, 1 billion ha of natural lands will be converted to agriculture (Tilman et al., 2001; Foley et al., 2005; Gibbs et al., 2010; Foley et al., 2011). Land clearance for agriculture is a major threat to biodiversity and ecosystem function (Chapin et al., 2000; Kremen, 2005; Tilman et al., 2001; Lobell and Field, 2007).

Conversion pressure is especially high in the tropics, which contain much of the world's plant and vertebrate diversity (Lambin et al., 2003; Gibson et al., 2011; Bhagwat et al., 2008). Conserving tropical forest depends on effectively managing human-modified landscapes. This could be achieved by adopting best practices methods for plantation management leading to more extensive agricultural areas, or more intensive agriculture with the aim of conserving larger tracts of natural forest.(Radford and Bennett, 2007; Lindenmayer et al., 2008; Gardner et al., 2009; Phalan et al., 2011; Kamp et al., 2015; Lamb et al., 2016).

Rubber (*Hevea brasiliensis*) is one of the most rapidly expanding cash crops in tropical Asia (Xu et al., 2005; Ziegler et al., 2009; Fox et al., 2012; Fox et al., 2014; Sahner et al., 2015). From 1983 to 2012, the area of planted rubber increased from 5.5 to 9.9 million ha, with much of that increase occurring in Southeast Asia and Southwest China (Fox et al., 2014; Warren-Thomas et al., 2015). The global area of planted rubber is projected to expand further over the next two decades (Fox and Castella, 2013; Zomer et al., 2014). The expansion of rubber across the range of planting intensities, from low-intensity jungle rubber where the crops are interspersed with native trees to high-intensity monoculture, has caused species losses, even in low-intensity mixed plantations (Thiollay, 1995; Peh et al., 2006; Beukema et al., 2007; Phommexay et al., 2011; Meng et al., 2012, Sreekar et al., 2016).

A major debate facing commodity plantation crops such as rubber is whether or not economically feasible methods can be adopted to render monoculture plantations more amenable for biodiversity conservation (Nájera and Simonetti, 2010; Laurance et al., 2010; Sreekar et al., 2016). It is also unclear if certain landscape- and plantation-level planting configurations produce better outcomes for native avian retention. For rubber, potential site-level interventions include leaving fruiting mistletoe (Sreekar et al., 2016), manipulating inter-tree planting distance, and creating more diverse microhabitats (Nájera and Simonetti, 2009). In addition, previous research has largely focused on areas where rubber is not planted by smallholders (Fox and Castella, 2013; Warren-Thomas et al., 2015), or has contrasted the impact of jungle rubber—rubber interspersed with native trees—against monocultures. Our study presents a novel analysis of the impacts of micro-

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scale habitat features in rubber plantation versus landscape scale natural forest retention in a region with intensive rubber monoculture.

Resolving pathways to increase the wildlife friendliness of rubber is particularly salient in tropical Southern China, where rubber offers much higher economic returns than traditional cultivation (Liu et al., 2006). In China, Hainan province and Xishuangbanna Dai Autonomous Prefecture within Yunnan Province (henceforth Xishuangbanna) have dominated rubber production (Xu et al., 2005; Guo et al., 2006). These two locations are also biodiversity hotspots and Xishuangbanna supports nationally and internationally important levels of diversity as it falls in the transition zone from the subtropics to the tropics (MacKinnon and MacKinnon, 1986; Myers et al., 2000; Guo et al., 2002; Li et al., 2007).

Despite containing only 0.2% of China's landmass, Xishuangbanna is home to more than one third of all the avian diversity in China (Zhang and Cao, 1995). In Xishuangbanna, rubber is mainly distributed under 900 m elevation ASL and the expansion of rubber has led to a decrease in regional biodiversity (Li et al., 2009; Chang et al., 2013; Yi et al., 2014) and ecosystem function (Zhang et al., 2003; Hu et al., 2008). Conversion to rubber is projected to intensify for the remaining native forest types, including some with high endemic richness such as karst outcroppings and montane rainforest (Xu, 2006; Chen et al., 2016). We examined bird diversity in the rubber-natural forest mosaic of Xishuangbanna. Birds are important seed dispersers (Corlett, 1998), have strong interactions with other vertebrate taxa (Blair, 1999; Schulze et al., 2004), are sensitive to environmental change, and can be easily observed (Melles et al., 2003; Chang et al., 2013). We investigated the following questions:

1) Which functional groups of birds can persist in rubber plantations?
2) How do changes at the plot level (vegetation structure and management style in a small area) affect bird diversity within rubber plantations?
3) How do bird species respond to changes in natural forest at landscape level?

2. Methods

2.1. Study site and regions

The study was conducted in Nabanhe nature reserve, Xishuangbanna, Yunnan province, China, located between 22°04′-22°17′N, 100°32′-100°44′E and totaling around 26,600 ha. We also included an adjacent state-owned farm, Mansha, that was devolved to villager smallholding in 2010. Nabanhe supports lowland and montane deciduous tropical rainforest. The elevational range of Nabanhe is 540–2300 m ASL with a mean annual temperature between 18°-22°, and overall precipitation totaling 1100–1600 mm. Xishuangbanna has two distinct seasons: a rainy season from May to October and a dry season from November to April of the next year (Zhang and Cao, 1995).

There are 31 villages within the administrative areas overseen by Nabanhe's protected area bureau. The main form of employment for local villagers is smallholder agriculture, primarily rubber plantations (Yang et al., 2006). The rubber plantations in these villages ranged from 1 to 130 ha in area in 2005 (Yang et al., 2006). Additional crop types grown in and around rubber plantations are described in the subsequent section "Environmental variables at plot and landscape scales".

Our study focused on two scales of habitat features: a landscape and individual plantation (plot) scale. From south to north along the Nabanhe river, the area of rubber plantation decreases with higher altitude. For the landscape level study, we chose 19 circular sampling regions (at a radius of 500 m) along the river to cover the gradient in rubber-nature forest composition and elevation change (Fig. 1). We calculated the coefficient of variation for the ratio of rubber to natural at radii of 500, 600, 700, 800, 900, and 1000 m; and 500 m was chosen to maximize sample size and the gradient in the ratio of rubber to forest across the sites. For the plot level, we chose a scale of 50 m-radius circular area, located within each of the 19 sampling regions, similar to previous avian impact assessments in tropical agriculture (Cushman and McGarigal, 2002; Peh et al., 2005).

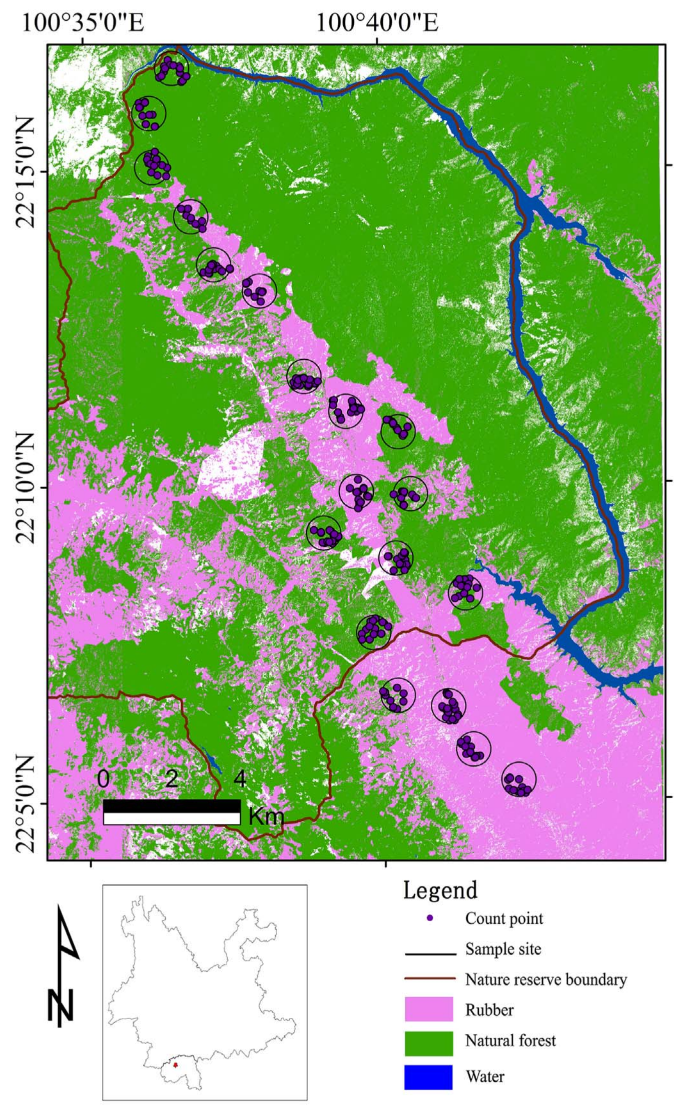

Fig. 1. Avian sampling sites and point count stations in Nabanhe nature reserve, Xishuangbanna, Yunnan, China. The sampling regions are marked with circles. The inset depicts where Nabanhe is located in relation to Xishuangbanna and Yunnan Province, China; and count points distribution within one region.

All study locations were limited to rubber planted 6 or more years ago; intensive management such as frequent use of herbicide or insecticide, only commences upon rubber tapping, which begins when plants reach 6–7 years of age.

2.2. Bird sampling

Birds were sampled using repeated point counts. Points were situated along trails and both rubber and natural forest were surveyed. Each point was located at least 200 m away from other points to ensure quasi-independence (Ralph et al., 1995). We surveyed each point for 8 min and all birds seen or heard within a 50 m radius were recorded. All surveys were performed by the lead author, who has more than a decade of bird watching experience in Southern Yunnan. One difficult group is Phylloscopus, which was identified mostly by call or song, including migratory species. In total, > 70% of the Phylloscopus species can be identified to species level. Flyovers and raptors were excluded from analysis. All points were surveyed within 3 h of sunrise. Each region was surveyed twice during both the wet and dry seasons, and 15 point count stations were established in each of the 19 sampling regions, totaling 285 distinct points. The survey was performed from March 2013 to May 2015.

2.3. Environmental variables at plot and landscape scales

Nearly all rubber planted in Xishuangbanna are smallholder plantations with planting practices varying between individual farmers. Some plantations have large inter-tree gaps with a density of about 300 trees/ha, typically a legacy from the previous period of state-owned collective farms which were then devolved to private ownership (Huang et al., 2015). Native trees are occasionally retained within rubber plantations. Some smallholders grow tea (*Camellia* spp.) or maintain a small area of cropland (e.g. corn, sugarcane) or a fishing pool under the rubber canopy. However, most farmers plant rubber at high densities and exhaustively clear all undergrowth.

To collect environmental variables at plot scale within rubber, we measured the mean distance between a grid of 5 × 5 rubber trees in each plot. We also collected vegetation and land use data within a 50-m circular area around each count point to measure canopy cover (low, middle, high), presence of water (1 or 0), the age of the rubber, grass coverage (0–80%), number of native trees, and the number of distinct land usages (presence or absence of fishing pool and cropland). The border of the plot was measured by a laser rangefinder (S9 Rangefinder, Shenzhen MileSeey Technology Co., Ltd.). The inter-tree planting distances at each plot were measured by a single observer and all other metrics were estimated by the lead author to reduce observer bias. We also calculated forest area at the plot scale from raster data using a F_{IDW} (inverse distance weight) $_{metric}$ at radii of 200 m, 400 m and 800 m around each plot (Edwards et al., 2014).

The land cover types in the greater study area were interpreted from Spot 6 images acquired in March 2014. We classified land cover type into rubber, natural forest, water, and developed area. The interpretation was performed using the supervised classification function in a trial version of Erdas Imagine 9.2; incorrectly classified locations were then eye-interpreted and manually modified. Landscape level metrics in study regions were measured using the FRAGSTATS software (McGarigal et al., 2002). The landscape metrics included the area of natural forest, distance to the nearest large (> 100 ha) forest patch, and the percentage of natural forest within each region, calculated at a 500 m–radius circular area.

2.4. Statistical analysis

All analyses were performed in R (v. 3.2.1) using the packages ‘vegan’ (Oksanen et al., 2007), ‘lme4’ (Bates et al., 2014), ‘ape’ (Kühn and Dormann, 2012), and ‘MuMIn’ (Bartoń, 2013). Residual spatial autocorrelation was assessed using Moran's I for the global landscape-scale models for each category of bird species (all species, forest-dependents, and the community-wide NMDS analysis) (Kühn and Dormann, 2012).

To compare the richness of birds in rubber and natural forest, individual-based rarefaction was conducted. To examine changes in community composition between rubber and natural forest locations, we evaluated the number of bird species found in rubber, natural forest, and both land cover types. The bird species were classified to feeding guilds (nectivores, granivores, carnivores, insectivores and frugivores) and as forest-dependent or non-forest dependent based on published sources (MacKinnon et al., 2000; Yang et al., 2004; Del Hoyo et al., 2017). The response of these functional guilds to plot and landscape level change were tested. As richness patterns alone are insufficient to determine trends in community composition (Barlow et al., 2010; Solar et al., 2015), we also performed Non-metric Multi Dimensional Scaling (NMDS) and used this metric as a regression response variable.

At the landscape level (radius = 500 m), the environmental variables were natural forest area (ha) and distance to large (> 100 ha) forest patches (m). The plot-level variables were the age of rubber tree (in years), mean rubber inter-tree distance (meters), the number of land usages (1–3), the number of native trees, percent coverage of grass, and canopy cover (binomial: 1 = high canopy cover) at 50 m circular area, $F_{IDW,}$ as defined above, and distance of each count point to the nearest large (> 100 ha) forest patch (m). For point level analyses of overall avian species richness, a random intercept for each landscape region (radius = 500 m) was included.

Continuous predictor variables were standardized to have a mean of 0 and a standard deviation of 1 (Gelman and Hill, 2006). Categorical variables were untransformed. Model and model-averaged equations are reported in terms of the standardized predictor variables for continuous variables and on the original scale for categorical variables. We used Generalized Linear Models (GLM) to examine the impact of environmental variables on overall avian species richness and functional group richness at landscape and plot scales with Poisson errors for overall, feeding guild, and forest-dependent richness, while the community ordination models used a Gaussian error distribution.

To determine the impact of each variable on these response variables, the set of predictor variables was dredged in R. Multi-model inference was performed for the best set of candidate models, which were determined by a cutoff threshold of $\Delta AIC_c < 2$ (Burnham and Anderson, 2004).

3. Results

3.1. Overall richness and community composition patterns

A total of 120 species of birds were recorded in the study area (see Appendix A), of which 108 were observed in natural forest and 59 in rubber. The individual-based rarefaction curve for the plot-level data indicated reasonable representation for both the rubber and natural forest habitats (Fig. 2), and demonstrated that landscapes where natural forest dominated supported significantly more avian richness.

Across all the sampled plots, the Chao2 asymptotic richness for each habitat type was 184.3 ± 30.8 species (natural forest) and 80.8 ± 12.7 species (rubber). Considering the full suite of birds found within the study area, the number of avian species that were solely found in forest was 61 (51%), 12 solely in rubber (10%), and 47 in both (39%). NMDS indicated that the natural forest and rubber communities were largely distinct (Fig. 3.); the asymptotic stress was 0.17, indicating that the top two dimensions explained a high percentage of variation.

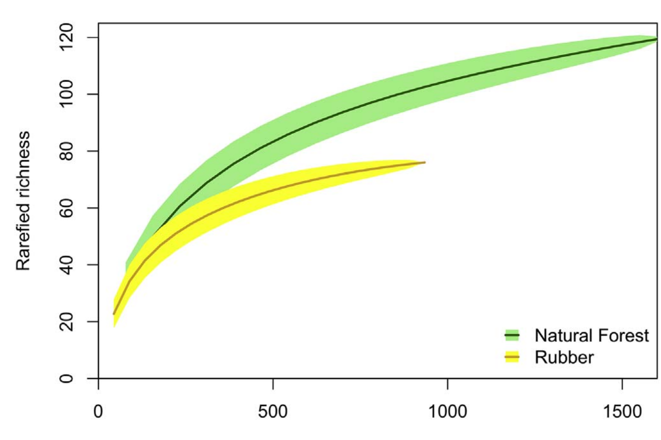

Fig. 2. The abundance-based accumulation curve generated from individual-based rarefaction of bird species richness in the natural forest (NF) and rubber (RB) sites for Nabanhe, Xishuangbanna.

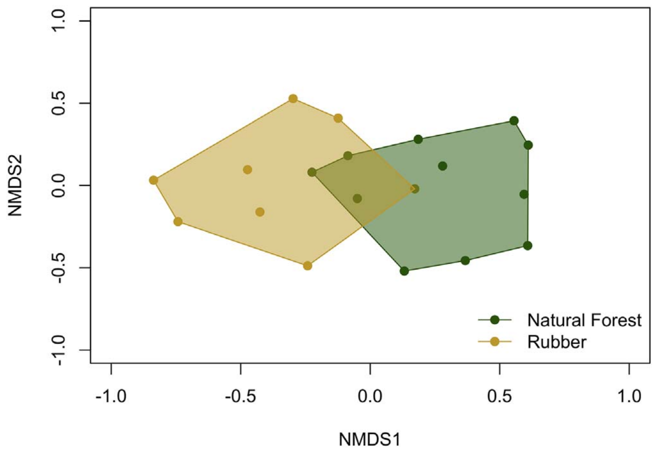

Fig. 3. Nonmetric multidimensional scaling (NMDS) ordination of species composition in natural forest and rubber plantation.

3.2. Birds guilds found in rubber

The most abundant functional groups in rubber plantations were insectivores (43 species) and frugivores (8 species). Other groups included carnivores (1 species: Greater coucal (*Centropus sinensis*), granivores (5species), and nectivores (2 species). Two species of water birds (*Bubulcus ibis* and *Egretta garzetta*) were seen perched on rubber trees nearby river banks.

3.3. Changes in bird diversity within rubber plantations

We collected environmental data around 45 count points with a 50 m circular radius. The overall avian richness at these points was modeled as the response variable. Multi-model inference indicated that the most supported model included rubber plantation age, the inter-tree planting distance, F_{IDW}, and Usage (Appendix B, Table A.1). The full model-averaged formula was:

$$\text{Log}(SPECIES\ RICHNESS) = 1.9 + 0.09 \times RUBBER\ PLANTATION\ AGE + 0.08 \times INTER\text{-}TREE\ DISTANCE – 0.03\ DISTANCE\ TO\ FOREST\ PATCH + 0.03 \times F_{IDW+}0.01\ USAGE$$

Species richness increases with rubber plantation age, greater inter-tree planting distance, more usages of the land and higher F_{IDW} (Fig. 4). The model-averaged predictions indicate that increase in age contribute mostly to species richness increase.

Variable importance calculated via Akaike weights indicated that distance to large forest patches was the most important variable (wdistolp = 0.01, wfidw = 0.002, $w_{age} = 0.008$, $w_{dist} = 0.008$). Among the variables, the age and inter-tree distance are features that land owners can manipulate directly. ANOVA and post-hoc Tukey's Honest Significant Difference with familywise error correction was performed for each of the variables to assess the impact of each predictor (at the levels of −1, 0, and 1 standard deviation from the mean, while all other variables were held constant at the mean value).

Plot-level richness was significantly different with age ($F\_2,131 = 10.7$, $p < 5 \times 10^{-5}$); post-hoc Tukey's Honest Significant Difference indicated that the mean richness across all three standardized age scores were significantly different. Richness was significantly different across the different inter-tree distance levels ($F\_2,131 = 7.166$, $p = 0.001$), but the Tukey's HSD indicated that only the mean richness at levels −1 and 1 of the inter-tree distance prediction were significantly different. There was no indication that plot-level richness changed significantly with f_IDW ($F\_2,131 = 0.25$, $p = 0.8$) or distance to the nearest large forest patch ($F\_2,131 = 0.7$, $p = 0.5$).

3.4. Landscape feature effects on bird diversity

3.4.1. Overall avian richness

Landscape level data were collected at 19 regions where avian species richness was summed across rubber and natural forest patches contained within each site. The set of predictor variables were natural forest area (range: 3 ha – 76 ha), distance to nearest large forest fragment (in km, range: 0 km - 1.5 km), all of which were centered and scaled before analysis. There was no evidence of significant residual spatial autocorrelation (Moran's $I = 0.049$, $s = 0.059$, $p = 0.077$).

Among all possible combinations of predictor variables, the most parsimonious model had an AICc of 498.4. The next best model had $\Delta AIC_c = 6.54$ (Appendix B, Table A.2). The most supported model contained both natural forest area and distance to nearest large forest patch:

$$\text{Log}(SPECIES\ RICHNESS) = 4.3 + 0.14\ DISTANCE\ TO\ FOREST + 0.47\ NATURAL\ FOREST\ AREA$$

Our model predictions indicated that gains in natural forest area were associated with much higher levels of avian richness (Fig. 5 a). None of the landscapes were completely deforested. The sites below the 25% quantile for percent natural forest cover had a mean of 7.9 ± 0.9% forest cover, and a mean of 55 ± 2.6 species. Above the 75% quantile for natural forest cover, the mean percent cover rose to 93.9 ± 0.7%, and average species richness to 135.8 ± 18.8.

3.4.2. Forest dependent species

With the forest dependent species, the most parsimonious model had an AICc of 227.09 (Table A.3). The most parsimonious averaged model contained natural forest area and distance to nearest large forest patch:

$$\text{Log}(FOREST\ DEPENDENT\ SPECIES\ RICHNESS) = 4 + 0.24\ NATURAL\ FOREST\ AREA - 0.03\ DISTANCE\ TO\ FOREST$$

As with overall richness, larger areas of natural forest led to higher predicted levels of forest-dependent richness (Fig. 6).

3.4.3. Community composition

Using the ordination scores as a response variable, the most parsimonious GLM model had an AICc of 19.28. The next best model had $\Delta AIC_c = 1.11$ (Appendix C, Table A.4). The full model-average yielded a relationship of:

$$NMDS\ ORDINATION\ DISTANCE = -7 \times 10^{-18} + 0.4\ NATURAL\ FOREST\ AREA – 0.06\ DISTANCE\ TO\ FOREST$$

The distance of NMDS ordination increased linearly with the increase of Natural forest area.(Fig. 7).

3.4.4. Guild level patterns in landscape sites

Five feeding guilds of birds were recorded in the landscape level data: carnivores, frugivores, granivores, insectivores, nectarivores. Insectivore and frugivore richness was higher with larger natural forest area (Fig. 5 b). The other 3 feeding guilds did not evince a clear pattern with respect to natural forest area, but this was likely because these guilds were relatively depauperate across the study region. For these 3 guilds, the total number of species in each guild ranged from 5 (carnivores) to 7 (granivores).

4. Discussion

Our findings reiterate the irreplaceability of natural forests and indicate that agricultural lands, particularly monocultures, have limited conservation value in comparison to native tropical habitat (Koh, 2008;

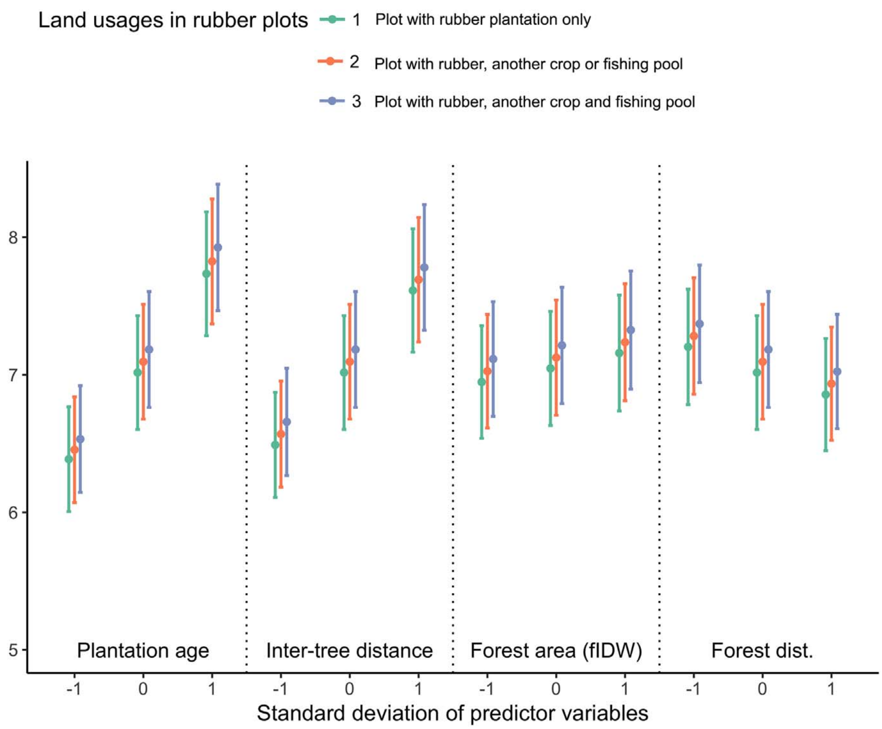

Fig. 4. The impact of landscape (F_{IDW}) vs. plantation features in determining species richness within rubber plots. The data were shown with − 1.0, + 1 stand deviation, the green means plot with rubber only, red means plot with rubber and another crop or fishing pool; the blue means plot with rubber, another crop and fishing pool. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

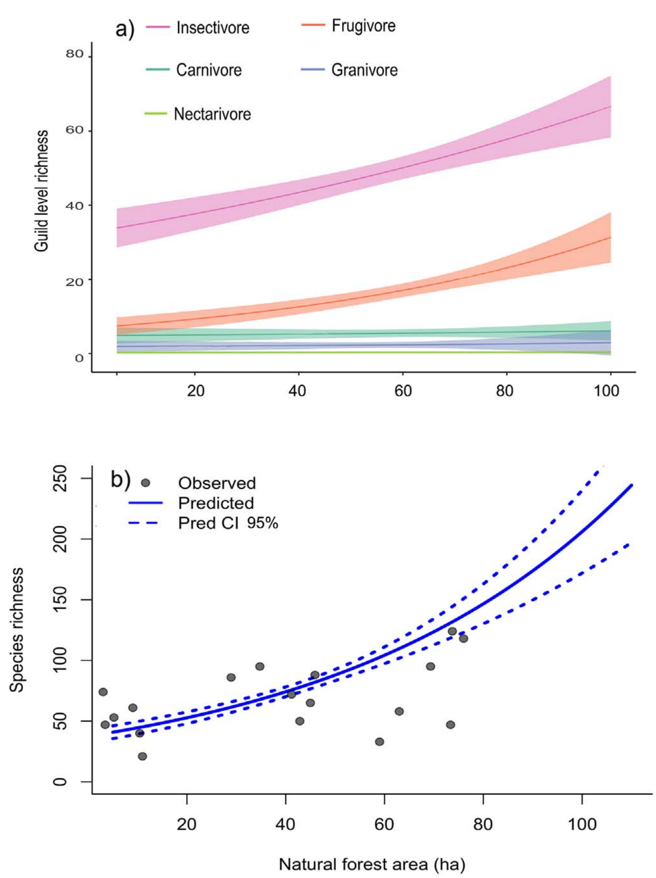

Fig. 5. a) Overall avian richness and b) different functional guilds were projected to increase in landscape regions with larger natural forest fragment area.

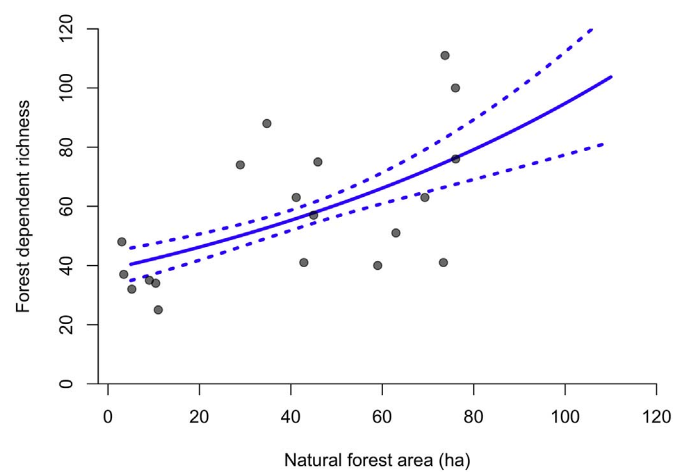

Fig. 6. The Forest dependent species richness was projected to increase with larger forest area in landscape regions.

Gibson et al., 2011). We observed substantial reductions in species richness in rubber as compared to natural forest, which is consistent with previous findings in intensive rubber monocultures (Aratrakorn et al., 2006; Peh et al., 2006; Sreekar et al., 2016). Among the species recorded, 73% were forest-dependent and only 27% exhibited some degree of flexibility in their habitat requirements. Rubber cannot support the majority of the tropical avifauna in our study region. Species that were solely observed in rubber were typically cosmopolitan generalists (e.g. Scaly-breasted munia *Lonchura punctulata* and Oriental Magpie Robin *Copsychus saularis*), which have low conservation value.

However, there are beneficial, though limited, pathways for rendering rubber more hospitable toward wildlife. The first is retaining older trees, and the second is increasing inter-tree planting distance. There is some degree of interaction in this observation, as almost all of the older plantations (those where the trees are 30 or more years old), were originally state-owned and planted with gap distances larger than that

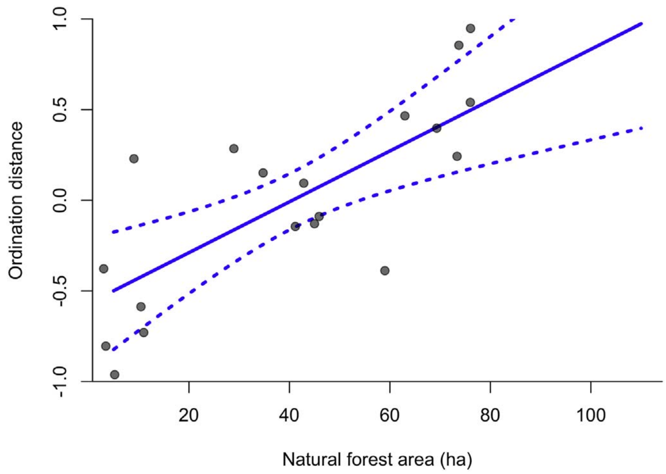

Fig. 7. The NMDS ordination was projected to increase with larger forest area in landscape regions.

of the typical smallholder today (Xu, 2006; Yi et al., 2014). Older and larger trees can support more vines suitable for frugivore and insectivore foraging (Sreekar et al., 2016).

Larger inter-tree gaps allow more native elements to provide habitat for insects and small animals, which present attractive foraging opportunities. In addition, larger planting gaps could be a win-win for biodiversity and farmers; rubber grown at a density of around 450 tree/ha (a bit higher than the density we observed in the study area, but still lower than the density of smallholder plantation) will ensure more stable latex production (Huang et al., 2015). Intercropping pineapple (*Ananas comosus*), tea (*Camellia sinensis*), pepper (*Piper* spp.), and medicinal plants (e.g. *Alpinia oxyphylla*) will increase the production of latex, soil fertility and reduce soil erosion within the rubber plantation (Long, 1991; Lin et al., 1999). Increasing inter-tree distance, decreasing the frequency of rubber tree replanting, and retaining native shrubs, or intercrop some other crops may present low-cost methods for improving rubber plantations for avian populations and farmer well-being.

To the authors' knowledge, this is the first study implemented at a landscape scale in rubber-forest mosaic. Our results showed that increasing forest area would exert the most dramatic impacts on local and landscape-level avian diversity. Even a relatively modest increase in forest area would yield large gains in species richness, this may also due to the improve of forest quality as larger forest patch would be better in protecting biodiversity than in ‘spared’ forest (Edwards et al., 2010), as well as help reduce the erosion of forest quality and associated loss of biodiversity (Barlow et al., 2016 Nature).

Distance to large forest patches was one of the variables included after multi-model comparison, gesturing to the importance of forests as reservoirs of avian diversity. As such, a priority for rubber-producing regions must be increasing the overall area of natural forest conserved, as well as enhancing their connectivity.

At the landscape scale, frugivores and insectivores exhibited the most marked response to forest area changes. Chang et al. (2013) found that ground insectivores, frugivores, and large-bodied birds were the most sensitive groups to forest fragmentation in Xishuangbanna. These findings are similar, suggesting that frugivores and insectivores may be particularly useful sentinel groups for assessing the success of forest conservation (Turner and Corlett, 1998; Şekercioğlu et al., 2002; Sodhi et al., 2004; Şekercioğlu, 2012).

These findings speak to the land sharing-sparing debate, and further demonstrate the limited value of farmland improvements versus forest conservation (Aratrakorn et al., 2006; Phalan et al., 2011; Lee et al., 2014). Our results clarify the impact of rubber-natural forest mosaic landscapes on avian diversity. Future research can and should integrate socioeconomic data, and consider impacts of farm- and broad-scale interventions on yield (Fischer et al., 2008; Chandler et al., 2013; Grau et al., 2013; Gilroy et al., 2014; Kremen, 2015). Although fine-scale, farm-level interventions could render rubber plantations more hospitable to native avian diversity, our findings stress that land sparing must be implemented for successful conservation in rubber-dominated landscapes (Lee et al., 2014; Edwards et al., 2015). Alarmingly, not only do forecasts predict that forest clearance for commodity agriculture will continue in this landscape (Xu et al., 2014), but there have also been observations of deforestation within the bounds of Nature Reserves (Zhang and Cao, 1995). Given recent interest by the prefecture government of Xishuangbanna to promote eco-friendly rubber plantation practices (Zhang, 2015), our results strongly indicate that the main focus of any such intervention must emphasize conserving large and intact forest patches.

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Appendix A. The species observed in rubber plantation and natural forest in Nabanhe, Xishuangbanna, Yunnan

Bird species	Functional guild[a]	Occurance[b]	Forest dependent[c]
Abroscopus superciliaris	I	Both	P
Aegithina tiphia	I	NF	NP
Aethopyga siparaja	N	NF	NP
Alcedo atthis	I	R	NP
Alcippe dubia	I	NF	P
Alcippe morrisonia	I	NF	P
Alcippe poioicephala	F	Both	P
Alophoixus pallidus	N	Both	P
Anthreptes malacensis	N	NF	P

Anthreptes singalensis	N	NF	P
Apus affinis	I	Both	NP
Arachnothera longirostris	I	Both	P
Arachnothera magna	N	NF	P
Blythipicus pyrrhotis	I	NF	P
Bubulcus ibis	I	R	NP
Cacomantis merulinus	I	Both	P
Cacomantis sonneratii	C	NF	P
Carpodacus erythrinus	G	R	P
Centropus sinensis	C	Both	NP
Cettia acanthizoides	I	R	NP
Chaimarrornis leucocephalus	F	R	NP
Chloropsis cochinchinensis	I	Both	P
Chloropsis hardwickii	I	NF	P
Copsychus malabaricus	I	Both	NP
Copsychus saularis	I	R	NP
Cuculus micropterus	I	Both	P
Cuculus poliocephalus	I	NF	P
Culicicapa ceylonensis	I	Both	P
Cyornis banyumas	I	Both	P
Dendrocopos canicapillus	I	NF	P
Dicaeum chrysorrheum	N	NF	P
Dicaeum concolor	N	Both	P
Dicaeum cruentatum	N	NF	P
Dicrurus aeneus	I	NF	P
Dicrurus annectans	I	NF	P
Dicrurus hottentottus	I	Both	P
Dicrurus leucophaeus	I	Both	P
Dicrurus macrocercus	I	Both	P
Dicrurus remifer	I	R	P
Egretta garzetta	I	Both	NP
Emberiza rutila	G	NF	NP
Eudynamys scolopacea	I	Both	NP
Eurystomus orientalis	I	NF	P
Ficedula parva	I	R	P
Gallus gallus	I	NF	P
Garrulax chinensis	F	NF	P
Garrulax leucolophus	C	NF	P
Glaucidium brodiei	C	NF	P
Glaucidium cuculoides	I	Both	P
Harpactes erythrocephalus	I	R	P
Hemiprocne coronata	I	R	P
Hemipus picatus	F	Both	P
Hemixos flavala	I	NF	P
Hirundo rustica	I	Both	NP
Hypothymis azurea	I	NF	P
Hypsipetes leucocephalus	F	NF	P
Hypsipetes mcclellandii	F	NF	P
Iole propinqua	G	Both	P
Lanius schach	I	NF	NP
Lonchura punctulata	G	R	NP
Lonchura striata	G	Both	NP
Lophura nycthemera	G	NF	NP
Luscinia calliope	I	NF	NP
Macronous gularis	F	Both	P
Megalaima asiatica	F	Both	P
Megalaima australis	F	NF	P
Megalaima haemacephala	F	NF	P
Megalaima virens	I	Both	P
Melanochlora sultanea	I	NF	P
Mesophoyx intermedia	C	NF	NP
Motacilla alba	I	NF	NP
Motacilla citreola	I	NF	NP
Niltava macgrigoriae	I	NF	P
Niltava vivida	I	NF	P
Nyctyornis athertoni	I	NF	P

Oriolus traillii	I	NF	P
Orthotomus atrogularis	I	NF	P
Orthotomus sutorius	I	Both	NP
Parus major	I	Both	P
Pellorneum albiventre	I	NF	NP
Pellorneum ruficeps	I	Both	P
Pellorneum tickelli	I	Both	NP
Pericrocotus brevirostris	I	NF	P
Pericrocotus flammeus	I	NF	P
Phaenicophaeus tristis	I	Both	P
Phylloscopus inornatus	I	Both	P
Phylloscopus reguloides	I	Both	P
Picumnus innominatus	I	NF	P
Picus chlorolophus	I	R	P
Picus flavinucha	I	NF	P
Polyplectron bicalcaratum	I	NF	P
Pomatorhinus ochraceiceps	I	NF	P
Prinia flaviventris	I	NF	NP
Prinia hodgsonii	I	Both	NP
Prinia inornata	I	NF	NP
Prinia rufescens	F	Both	NP
Psittacula finschii	F	NF	P
Pteruthius flaviscapis	I	NF	NP
Pteruthius xanthochlorus obscurus	F	NF	P
Pycnonotus aurigaster	F	Both	NP
Pycnonotus jocosus	F	Both	NP
Pycnonotus melanicterus	I	Both	P
Rhipidura albicollis	I	Both	P
Rhipidura hypoxantha	I	NF	P
Sasia ochracea	I	NF	P
Seicercus poliogenys	I	Both	P
Serilophus lunatus	I	NF	P
Sitta castanea	I	Both	P
Sitta frontalis	I	NF	P
Stachyris chrysaea	I	Both	P
Stachyris ruficeps	I	Both	P
Stachyris striolata	I	NF	P
Surniculus lugubris	I	Both	P
Tephrodornis gularis	I	NF	P
Terpsiphone paradisi	I	NF	P
Urocissa erythrorhyncha	G	Both	P
Yuhina castaniceps	I	Both	P
Yuhina zantholeuca	I	NF	P
Zosterops japonicus	I	NF	P
Zosterops palpebrosus	I	Both	P

[a] I = insectivore; F = frugivore; G = granivore; C = carnivore; N = nectivore.
[b] Both = The species was found in both natural forest and rubber; N = natural forest; R = rubber.
[c] P = Prefer forest; NP = Not prefer forest.

Appendix B. Models fitting for landscape and plot level analyses

Table A.1
The most supported candidate models of plot-level covariates on avian diversity. Age is the age (years) of rubber plantation canopy means the canopy cover; Dishor represent the inter-tree distance; F_{IDW} is the sum of inverse distance weight to forest in 200 m, 400 m and 800 m buffer area."Grass" the percent grass cover at each plot; N_{trees} the number of native trees retained in each plot; and "Usages" the number of land uses within each plot.

	Intercept	Age	Dis_hor	distolp	FIDW	Usage	Df	logLik	AICc	delta	Weight
6	1.94	0.12	0.12				4	− 129.19	267.4	0.00	0.22
14	1.90	0.15	0.14	-0.15			5	− 128.21	267.9	0.56	0.17
2	1.93	0.11					3	− 130.75	268.1	0.69	0.16
5	1.93		0.11				3	− 130.86	268.3	0.93	0.14
22	1.83	0.13	0.13		0.25		5	− 128.53	268.6	1.22	0.12
1	1.92						2	− 132.17	268.6	1.25	0.12
130	1.78	0.14				0.12	4	− 130.10	269.2	1.81	0.09

Table A.2
Multi-model inference results for comparing candidate models of landscape-level covariates on avian diversity. The variables are distance to large forest patch (distolp), and natural forest area (NFarea).

	(Intercept)	distolp	NFarea	df	logLik	AICc	delta	weight
4	4.32	0.14	0.46	3	− 245.38	498.36	0	0.96
3	4.32		0.38	2	− 250.07	504.89	6.54	0.04
2	4.36	− 0.23		2	− 315.28	635.3	136.95	0[a]
1	4.38			1	− 345.54	693.31	194.96	0[a]

[a] The 0 means the number is smaller than 10^{-16}.

Table A.3
Multi-model inference results for comparing candidate models of landscape-level covariates on forest dependent avian diversity. The variables are distance to large forest patch (distolp), and natural forest area (NFarea).

	(Intercept)	distolp	NFarea	df	logLik	AICc	delta	weight
3	4.02		0.26	2	− 111.17	227.09	0[a]	0.62
4	4.02	− 0.07	0.22	3	− 110.25	228.1	1	0.38
2	4.03	− 0.24		2	− 121.77	248.3	21.21	0[a]
1	4.05			1	− 145.12	292.47	65.38	0[a]

[a] The 0 means the number is smaller than 10^{-16}.

Table A.4
Multi-model inference results for comparing candidate models of landscape-level covariates on NMDS ordination distance. The variables are distance to large forest patch (distolp), and natural forest area (NFarea).

	(Intercept)	distolp	NFarea	df	logLik	AICc	delta	weight
3	0[a]		0.43	3	− 5.84	19.28	0[a]	0.6
4	0[a]	− 0.16	0.31	4	− 4.77	20.39	1.11	0.34
2	0[a]	− 0.39		3	− 8.22	24.04	4.76	0.06
1	0[a]			2	− 14.86	34.46	15.18	0[a]

[a] The 0 means the number is smaller than 10^{-16}.

References

Aratrakorn, S., Thunhikorn, S., Donald, P.F., 2006. Changes in Bird Communities Following Conversion of Lowland Forest to Oil Palm and Rubber Plantations in Southern Thailand. Vol. 16. Bird Conservation Internationalpp. 71–82.

Barlow, J., Gardner, T.A., Louzada, J., Peres, C.A., 2010. Measuring the conservation value of tropical primary forests: the effect of occasional species on estimates of biodiversity uniqueness. PLoS One 5, e9609.

Barlow, J., Lennox, G.D., Ferreira, J., Berenguer, E., Lees, A.C., Mac Nally, R., Thomson, J.R., de Barros Ferraz, S.F., Louzada, J., Oliveira, V.H.F., 2016. Anthropogenic disturbance in tropical forests can double biodiversity loss from deforestation. Nature 535, 144–147.

Bartoń, K., 2013. MuMIn: multi-model inference. In R package version package version 1.15.6. https://CRAN.R-project.org/package=MuMIn.

Bates, D., Mächler, M., Bolker, B., Walker, S., 2014. Fitting linear mixed-effects models using lme4. J. Stat. Softw. 67, 1–48.

Beukema, H., Danielsen, F., Vincent, G.g., Hardiwinoto, S., Andel, J.v., 2007. Plant and bird diversity in rubber agroforests in the lowlands of Sumatra, Indonesia. Agrofor. Syst. 70, 217–242.

Bhagwat, S.A., Willis, K.J., Birks, H.J.B., Whittaker, R.J., 2008. Agroforestry: a refuge for tropical biodiversity? Trends Ecol. Evol. 23, 261–267.

Blair, R.B., 1999. Birds and butterflies along an urban gradient: surrogate taxa for assessing biodiversity? Ecol. Appl. 9, 164–170.

Burnham, K.P., Anderson, D.R., 2004. Multimodel inference understanding AIC and BIC in model selection. Sociol. Methods Res. 33, 261–304.

Chandler, R.B., King, D.I., Raudales, R., Trubey, R., Chandler, C., ChÁvez, V.J., 2013. A small-scale land-sparing approach to conserving biological diversity in tropical agricultural landscapes. Conserv. Biol. 27, 785–795.

Chang, X., Quan, R.-c., Wang, L., 2013. Bird conservation in extremely small tropical rainforest patches in southwest China. Biol. Conserv. 158, 188–195.

Chapin III, F.S., Zavaleta, E.S., Eviner, V.T., Naylor, R.L., Vitousek, P.M., Reynolds, H.L., Hooper, D.U., Lavorel, S., Sala, O.E., Hobbie, S.E., 2000. Consequences of changing biodiversity. Nature 405, 234–242.

Chen, H., Yi, Z.-F., Schmidt-Vogt, D., Ahrends, A., Beckschäfer, P., Kleinn, C., Ranjitkar, S., Xu, J., 2016. Pushing the limits: the pattern and dynamics of rubber monoculture expansion in Xishuangbanna, SW China. PLoS One 11, e0150062.

Corlett, R.T., 1998. Frugivory and seed dispersal by vertebrates in the Oriental (Indomalayan) Region. Biol. Rev. Camb. Philos. Soc. 73, 413–448.

Cushman, S., McGarigal, K., 2002. Hierarchical, multi-scale decomposition of species-environment relationships. Landsc. Ecol. 17, 637–646.

Defries, R.S., Rudel, Thomas, Uriarte, Maria, Hansen, M., 2010. Deforestation driven by urban population growth and agricultural trade in the twenty-first century. Nat. Geosci. 3, 178–181.

Del Hoyo, J., Elliot, A., Sargatal, J., Christie, D.A., 2017. Handbook of the Bird of the World. Lynx. http://www.hbw.com/.

Edwards, D.P., Magrach, A., Woodcock, P., Ji, Y., Lim, N.T.-L., Edwards, F.A., Larsen, T.H., Hsu, W.W., Benedick, S., Khen, C.V., 2014. Selective-logging and oil palm: multitaxon impacts, biodiversity indicators, and trade-offs for conservation planning. Ecol. Appl. 24, 2029–2049.

Edwards, D.P., Gilroy, J.J., Thomas, G.H., Uribe, C.A.M., Haugaasen, T., 2015. Land-sparing agriculture best protects avian phylogenetic diversity. Curr. Biol. 25, 2384–2391.

Edwards, D.P., Hodgson, J.A., Hamer, Keith C., Mitchell, S.L., Ahmad, A.H., Cornell, S.J., Wilcove, D.S., 2010. Wildlife-friendly oil palm plantations fail to protect biodiversity effectively. Conserv. Lett. 3, 236–242.

Fischer, J., Brosi, B., Daily, G.C., Ehrlich, P.R., Goldman, R., Goldstein, J., Lindenmayer, D.B., Manning, A.D., Mooney, H.A., Pejchar, L., Ranganathan, J., Tallis, H., 2008. Should agricultural policies encourage land sparing or wildlife-friendly farming? Front. Ecol. Environ. 6, 380–385.

Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E.A., Kucharik, C.J., Monfreda, C., Patz, J.A., Prentice, I.C., Ramankutty, N., Snyder, P.K., 2005. Global consequences of land use. Science 309, 570–574.

Foley, J.A., Ramankutty, N., Brauman, K.A., Cassidy, E.S., Gerber, J.S., Johnston, M., Mueller, N.D., O'Connell, C., Ray, D.K., West, P.C., Balzer, C., Bennett, E.M., Carpenter, S.R., Hill, J., Monfreda, C., Polasky, S., Rockström, J., Sheehan, J., Siebert, S., Tilman, D., Zaks, D.P.M., 2011. Solutions for a cultivated planet. Nature 478, 337–342.

Fox, J., Castella, J.-C., 2013. Expansion of rubber (*Hevea brasiliensis*) in Mainland Southeast Asia: what are the prospects for smallholders? J. Peasant Stud. 40, 155–170.

Fox, J., Castella, J.-C., Ziegler, A.D., 2014. Swidden, rubber and carbon: Can REDD+ work for people and the environment in Montane Mainland Southeast Asia? Glob. Environ. Chang. 29, 318–326.

Fox, J., Vogler, J.B., Sen, O.L., Giambelluca, T.W., Ziegler, A.D., 2012. Simulating land-

cover change in montane mainland southeast Asia. Environ. Manag. 49, 968–979.
Gardner, T.A., Barlow, J., Chazdon, R., Ewers, R.M., Harvey, C.A., Peres, C.A., Sodhi, N.S., 2009. Prospects for tropical forest biodiversity in a human-modified world. Ecol. Lett. 12, 561–582.
Gelman, A., Hill, J., 2006. Data Analysis Using Regression and Multilevel/Hierarchical Models. Cambridge University Press.
Gibbs, H.K., Ruesch, A.S., Achard, F., Clayton, M.K., Holmgren, P., Ramankutty, N., Foley, J.A., 2010. Tropical forests were the primary sources of new agricultural land in the 1980s and 1990s. Proc. Natl. Acad. Sci. U. S. A. 107, 16732–16737.
Gibson, L., Lee, T.M., Koh, L.P., Brook, B.W., Gardner, T.A., Barlow, J., Peres, C.A., Bradshaw, C.J.A., Laurance, W.F., Lovejoy, T.E., Sodhi, N.S., 2011. Primary forests are irreplaceable for sustaining tropical biodiversity. Nature 478, 378–381.
Gilroy, J.J., Woodcock, P., Edwards, F.A., Wheeler, C., Uribe, C.A.M., Haugaasen, T., Edward, D.P., 2014. Optimizing carbon storage and biodiversity protection in tropical agricultural landscapes. Glob. Chang. Biol. 20, 2162–2172.
Grau, R., Kuemmerle, T., Macchi, L., 2013. Beyond 'land sparing versus land sharing': environmental heterogeneity, globalization and the balance between agricultural production and nature conservation. Curr. Opin. Environ. Sustain. 5, 477–483.
Guo, Z., Zhang, Y., Deegen, P., Uibrig, H., 2006. Economic analyses of rubber and tea plantations and rubber-tea intercropping in Hainan, China. Agrofor. Syst. 66, 117–127.
Hu, H., Liu, W., Cao, M., 2008. Impact of land use and land cover changes on ecosystem services in Menglun, Xishuangbanna, Southwest China. Environ. Monit. Assess. 146, 147–156.
Huang, H., Zhang, W., Huang, H., 2015. The Management and Development of Rubber in Xishuangbanna. Enterprise with New and Advanced Technology of China. pp. 5–6.
Guo, H., Padoch, C., Coffey, K., Aiguo, C., Yongnenga, F., 2002. Economic development, land use and biodiversity change in the tropical mountains of Xishuangbanna, Yunnan, Southwest China. Environ. Sci. Pol. 5, 471–479.
Kamp, J., Urazaliev, R., Balmford, A., Donald, P.F., Green, R.E., Lamb, A.J., Phalan, B., 2015. Agricultural development and the conservation of avian biodiversity on the Eurasian steppes: a comparison of land-sparing and land-sharing approaches. J. Appl. Ecol. 52, 1578–1587.
Koh, L., 2008. Can oil palm plantations be made more hospitable for forest butterflies and birds? J. Appl. Ecol. 45, 1002–1009.
Kremen, C., 2005. Managing ecosystem services: what do we need to know about their ecology? Ecol. Lett. 8, 468–479.
Kremen, C., 2015. Reframing the land-sparing/land-sharing debate for biodiversity conservation. Ann. N. Y. Acad. Sci. 1335, 52–76.
Kühn, I., Dormann, C.F., 2012. Less than eight (and a half) misconceptions of spatial analysis. J. Biogeogr. 39 (5), 995–998.
Lamb, A., Balmford, A., Greena, R.E., Phalana, B., 2016. Towhat extent could edge effects and habitat fragmentation diminish the potential benefits of land sparing? Biol. Conserv. 195, 264–271.
Lambin, E.F., Geist, H.J., Lepers, E., 2003. Dynamics of land-use and land-cover change in tropical regions. Annu. Rev. Environ. Resour. 28, 205–241.
Laurance, W.F., Koh, L.P., Butler, R., Sodhi, N.S., Bradshaw, C.J.A., Neidel, D., Consunji, H., Vega, J.M., 2010. Improving the performance of the roundtable on sustainable palm oil for nature conservation. Conserv. Biol. 24, 377–381.
Lee, J.S.H., Garcia-Ulloa, J., Ghazoul, J., Obidzinski, K., Koh, L.P., 2014. Modelling environmental and socio-economic trade-offs associated with land-sparing and land-sharing approaches to oil palm expansion. J. Appl. Ecol. 51, 1366–1377.
Li, H., Aide, M., Ma, Y., Liu, W., Cao, M., 2007. Demand for rubber is causing the loss of high diversity rain forest in SW China. Biodivers. Conserv. 16, 1731–1745.
Li, H., Ma, Y., Liu, W., Liu, W., 2009. Clearance and fragmentation of tropical rain forest in Xishuangbanna, SW, China. Biodivers. Conserv. 18, 3421–3440.
Lin, W., Zhou, Z., Huang, S., 1999. A review and Prospect of intercropping in rubber plantation in China. Chin. J. Ecol. 18, 43–52.
Lindenmayer, D., Hobbs, R.J., Montague-Drake, R., Alexandra, J., Bennett, A., Burgman, M., Cale, P., Calhoun, A., Cramer, V., Cullen, P., 2008. A checklist for ecological management of landscapes for conservation. Ecol. Lett. 11, 78–91.
Liu, W., Hu, H., Ma, Y., Li, H., 2006. Environmental and socioeconomic impacts of increasing rubber plantations in Menglun township, Southwest China. Mt. Res. Dev. 26, 245–253.
Lobell, D.B., Field, C.B., 2007. Global scale climate–crop yield relationships and the impacts of recent warming. Environ. Res. Lett. 2, 014002.
Long, Y., 1991. Evaluation of ecological and economical benefits of the man-made rubber-tea community. Chin. J. Ecol. 10, 37–40.
MacKinnon, J., MacKinnon, K., 1986. Review of the Protected Areas System in the Indo-Malayan Realm. IUCN, Gland, Switzerland.
MacKinnon, J.R., Phillipps, K., He, F., 2000. A Field Guide to the Birds of China. Oxford University Press, USA.
McGarigal, K., Ene, E., Holmes, C., 2002. FRAGSTATS (version 3): FRAGSTATS metrics. University of Massachusetts-produced program. Available at the following website:. http://www.umass.edu/landeco/research/fragstats/documents/fragstats documents.html.
Melles, S., Glenn, S., Martin, K., 2003. Urban bird diversity and landscape complexity: species-environment associations along a multiscale habitat gradient. Conserv. Ecol. 7, 5.
Meng, L.-Z., Martin, K., Weigel, A., Liu, J.-X., 2012. Impact of rubber plantation on carabid beetle communities and species distribution in a changing tropical landscape (southern Yunnan, China). J. Insect Conserv. 16, 423–432.
Myers, N., Mittermeier, R.A., Mittermeier, C.G., Da Fonseca, G.A., Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature 403, 853–858.
Nájera, A., Simonetti, J., 2009. Enhancing avifauna in commercial plantations. Conserv. Biol. 24, 319–324.
Nájera, A., Simonetti, J.A., 2010. Can oil palm plantations become bird friendly? Agrofor. Syst. 80, 203–209.
Oksanen, J., Kindt, R., Legendre, P., O'Hara, B., Stevens, M.H.H., Oksanen, M.J., Suggests, M., 2007. Vegan, in community ecology package. R package version 2.4-1. https://CRAN.R-project.org/package=vegan.
Peh, K.S.H., Sodhi, N.S., De Jong, J., Sekercioglu, C.H., Yap, C.A.M., Lim, S.L.H., 2006. Conservation value of degraded habitats for forest birds in southern Peninsular Malaysia. Divers. Distrib. 12, 572–581.
Peh, K.S.-H., Jong, J.d., Sodhi, N.S., Lim, S.L.-H., Yap, C.A.-M., 2005. Lowland rainforest avifauna and human disturbance: persistence of primary forest birds in selectively logged forests and mixed-rural habitats of southern Peninsular Malaysia. Biol. Conserv. 123, 489–505.
Phalan, B., Onial, M., Balmford, A., Green, R.E., 2011. Reconciling food production and biodiversity conservation: land sharing and land sparing compared. Science 333, 1289–1291.
Phommexay, P., Satasook, C., Bates, P., Pearch, M., Bumrungsri, S., 2011. The impact of rubber plantations on the diversity and activity of understorey insectivorous bats in southern Thailand. Biodivers. Conserv. 20, 1441–1456.
Radford, J.Q., Bennett, A.F., 2007. The relative importance of landscape properties for woodland birds in agricultural environments. J. Appl. Ecol. 44, 737–747.
Ralph, C.J., Sauer, J.R., Droege, S., 1995. Monitoring Bird Populations by Point Counts .
Sahner, J., Budi, S.W., Barus, H., Edy, N., Meyer, M., Corre, M.D., Polle, A., 2015. Degradation of root community traits as indicator for transformation of tropical lowland rain forests into oil palm and rubber plantations. PLoS One 10, 1–19.
Şekercioğlu, ÇH., Ehrlich, P.R., Daily, G.C., Aygen, D., Goehring, D., Sandí, R.F., 2002. Disappearance of insectivorous birds from tropical forest fragments. Proc. Natl. Acad. Sci. 99, 263–267.
Şekercioğlu, C.H., 2012. Bird functional diversity and ecosystem services in tropical forests, agroforests and agricultural areas. J. Ornithol. 153, 153–161.
Schulze, C.H., Waltert, M., Kessler, P.J., Pitopang, R., Veddeler, D., Mühlenberg, M., Gradstein, S.R., Leuschner, C., Steffan-Dewenter, I., Tscharntke, T., 2004. Biodiversity indicator groups of tropical land-use systems: comparing plants, birds, and insects. Ecol. Appl. 14, 1321–1333.
Sodhi, N.S., Liow, L., Bazzaz, F., 2004. Avian extinctions from tropical and subtropical forests. Annu. Rev. Ecol. Evol. Syst. 35, 323–345.
Solar, R.R.d.C., Barlow, J., Ferreira, J., Berenguer, E., Lees, A.C., Thomson, J.R., Louzada, J., Maués, M., Moura, N.G., Oliveira, V.H., 2015. How pervasive is biotic homogenization in human-modified tropical forest landscapes? Ecol. Lett. 18, 1108–1118.
Sreekar, R., Huang, G., Yasuda, M., Quan, R.-C., Goodale, E., Corlett, R.T., Tomlinson, K.W., 2016. Effects of forests, roads and mistletoe on bird diversity in monoculture rubber plantations. Sci. Rep. 6.
Thiollay, J.-M., 1995. The role of traditional agroforests in the conservation of rain forest bird diversity in Sumatra. Conserv. Biol. 9, 335–353.
Tilman, D., Clark, M., 2014. Global diets link environmental sustainability and human health. Nature 515, 518–522.
Tilman, D., Fargione, J., Wolff, B., D'Antonio, C., Dobson, A., Howarth, R., Schindler, D., Schlesinger, W.H., Simberloff, D., Swackhamer, D., 2001. Forecasting agriculturally driven global environmental change. Science 292, 281–284.
Warren-Thomas, E., Dolman, P.M., Edwards, D.P., 2015. Increasing demand for natural rubber necessitates a robust sustainability initiative to mitigate impacts on tropical biodiversity. Conserv. Lett. 8, 230–241.
Xu, J., Fox, J., Vogler, J.B., Yongshou, Z.P.F., Lixin, Y., Jie, Q., Leisz, S., 2005. Land-use and land-cover change and farmer vulnerability in Xishuangbanna prefecture in southwestern China. Environ. Manag. 36, 404–413.
Xu, J., 2006. The political, social, and ecological transformation of a landscape. Mt. Res. Dev. 26, 254–262.
Xu, J., Grumbine, R.E., Beckschäfer, P., 2014. Landscape transformation through the use of ecological and socioeconomic indicators in Xishuangbanna, Southwest China, Mekong Region. Ecol. Indic. 36, 749–756.
Yang, L., Wen, X., Han, L., Yang, X., 2004. Yunnan Avifauna. Yunnan Technology and Science Press, Yunnan.
Yang, Y., Wu, Z., Wang, D., 2006. Nabanhe Watershed National Nature Reserve, Xishuangbanna (in Chinese). Yunnan Science Press, Kunming, Kunming, Yunnan.
Yi, Z.-F., Cannon, C.H., Chen, J., Ye, C.-X., Swetnam, R.D., 2014. Developing indicators of economic value and biodiversity loss for rubber plantations in Xishuangbanna, southwest China: a case study from Menglun township. Ecol. Indic. 36, 788–797.
Zhang, J., 2015. Research on the conservation of the environment friendly ecological rubber plantation, Xishuangbanna. J. Yunnan Agric. Univ. 9, 24–29.
Zhang, J., Cao, M., 1995. Tropical forest vegetation of Xishuangbanna, SW China and its secondary changes, with special reference to some problems in local nature conservation. Biol. Conserv. 73, 229–238.
Zhang, Y., Xin, W., Yujie, W., Wenjie, L., Yuhong, Liu, 2003. Comparison research on hydrological effect of the canopy of the tropical seasonal rainforest and rubber forest in Xishuangbanna, Yunnan. Acta Ecol. Sin. 23, 2653–2665.
Ziegler, A.D., Fox, J.M., Xu, J., 2009. The rubber juggernaut. Science 324, 1024–1025.
Zomer, R.J., Trabucco, A., Wang, M., Lang, R., Chen, H., Metzgerd, M.J., Smajgle, A., Beckschäferf, P., Xu, J., 2014. Environmental stratification to model climate change impacts on biodiversity and rubber production in Xishuangbanna, Yunnan, China. Biol. Conserv. 170, 264–273.