Roles of fragmented and logged forests for bird communities in industrial *Acacia mangium* plantations in Indonesia

Motoko S. Fujita[1], Dewi M. Prawiradilaga[2], Tsuyoshi Yoshimura[3]

[1] Center for Southeast Asian Studies, Kyoto University, 46 Shimoadachi, Yoshida, Sakyo, Kyoto 606-8501, Japan; fujita@cseas.kyoto-u.ac.jp

[2] Research Center for Biology, Indonesian Institute of Sciences, Jl. Raya Bogor, Jakarta KM46, Cibinong 16911, Indonesia; dewi005@lipi.go.id

[3] Research Institute for Sustainable Humanosphere, Kyoto University, Gokasho, Uji, Kyoto 611-0011, Japan; tsuyoshi@rish.kyoto-u.ac.jp

Corresponding Author: Motoko S. Fujita

E-mail: fujita@cseas.kyoto-u.ac.jp

Tel/Fax: +81-757539645

Center for Southeast Asian Studies, Kyoto University, 46 Shimoadachi, Yoshida, Sakyo, Kyoto 606-8501, Japan

Abstract

Industrial timber plantations severely impact biodiversity in Southeast Asia. Forest fragments survive within plantations, but their conservation value in highly deforested landscapes in Southeast Asia is poorly understood. In this study, we compared bird assemblages in acacia plantations and fragmented forests in South Sumatra to evaluate each habitat's potential conservation value. To clarify the impact of habitat change, we also analyzed the response of feeding guild composition. Five habitat types were studied: large logged forest (LLF), burnt logged forest (BLF), remnant logged forest (RLF), 4-year-old acacia plantation (AP4), and 1-year-old acacia plantation (AP1). Estimated species richness (Chao 2) was highest in LLF then AP4 and BLF, while AP1 and RLF had lower estimated species richness. Community composition was roughly divided into two groups by non-metric multidimensional scaling ordination: acacia plantation and logged forest. Sallying substrate-gleaning insectivores, such as drongos, broadbills, and some flycatchers, were restricted to LLF, whereas acacia plantation hosted many terrestrial frugivores, such as doves. Although fragmented forests in our study site lacked several common tropical forest species, these fragments provide an important habitat for some sallying and terrestrial insectivores. A network of small riparian remnant forests could be a complementary habitat for some species, while the conservation value of burnt forest might be low. In conclusion, the highly fragmented forests in plantations are suboptimal habitats for birds

but are still very important, because large primary forest blocks have been nearly lost in the surrounding landscape.

Key Words

Acacia mangium, biodiversity, habitat fragmentation, landscape management, industrial timber plantations, Sumatra

Introduction

Tropical rain forests in Southeast Asia comprise some of the most biologically diverse ecosystems in the world (Sodhi and Brook 2006). Sundaland, which includes the islands of Sumatra, Borneo, Java, Bali, and the Malay Peninsula, is a significant hotspot with 139 endemic bird species (17% of the 815 species identified in this area), although only 7.8% of primary vegetation remains (Myers et al. 2000), and more is lost each year. The area is now experiencing a critical and rapid loss of biodiversity, mainly because of habitat changes due to forest degradation, deforestation, and overexploitation (Sodhi et al. 2004, 2009; Laurance 2007). Tropical rain forests are disappearing rapidly to logging and conversion into both farmland and large-scale industrial timber plantations of exotic species, such as acacia, rubber, and oil palm. In South and Southeast Asia, planted forests covered more than 25.5 million ha in 2010 and continue to expand in range at a rate of 0.58 million ha per year (FAO 2010).

Several studies have indicated that industrial timber plantations have severe impacts on biodiversity (Barlow et al. 2007b), but only a few have examined acacia plantations (soil macrofauna, Tsukamoto and Sabang 2005; beetles, Chung et al. 2000; mammals, Nasi et al. 2008 and McShea et al. 2009; birds, Styring et al. 2011). *Acacia mangium* is an important industrial tree species and grows rapidly even on wasteland (Cossalter and Pye-Smith 2003).

This species originated in New Guinea and Australia, where it grows in sparse woodlands with frequent fires. Due to its high adaptability and growth rate, *A. mangium* has been cultivated as a stable source of wood products and was introduced to South Sumatra province, where grasslands (*Imperata cylindrica*) have proliferated in the aftermath of logging and intensive rotational cultivation (Yokota and Inoue 1996). As a result, acacia is widely planted for pulp production by industrial concession companies in Indonesia and is displacing natural lowland forests, especially in Sumatra and Borneo. Given this rapid and extensive anthropogenic landscape alteration, there is an urgent need to clarify the impacts of habitat change on biodiversity and to construct an effective framework for landscape management in tropical regions (Gardner et al. 2009). Of particular importance is understanding how managed forests contribute to species diversity, because much of the landscape has already been altered.

One common feature of deforested and converted landscapes is forest fragmentation (Laurance and Laurance 1999; Lindenmayer et al. 2002). Forest fragmentation is known to decrease biodiversity in many tropical regions, and some authors submit that the conservation of small forest fragments is less important than large fragments (Beier et al. 2002; Edwards et al. 2010). Hill et al. (2011) showed that the bird community is highly nested in small fragments compared to that of insects, thus lowering the conservation value of small fragments for birds. Nevertheless, fragmented forests still have conservation value in tropical landscapes, where deforestation is proceeding at an alarming rate (Turner and Corlett 1996; Hawes et al. 2008; Struebig et al. 2008; McShea et al. 2009; Chang et al. 2013). In South Sumatra, fragmented natural vegetation occurs in a matrix of acacia plantations. As birds respond sensitively to differences in habitat conditions (Barlow et al. 2007a), evaluating habitat quality is essential to allocating conservation effort.

We focused on birds, which are considered indicator organisms for assessing the environment at the landscape scale (O'Connell et al. 2000). Birds have numerous ecosystem functions in pollination (Ricketts et al. 2004), pest control (Marquis and Whelan 1994), and seed dispersal (Wunderle Jr. 1997) that benefit human welfare and the economy, from local to global scales (Millennium Ecosystem Assessment 2005). As these functions are related to feeding guilds, analyzing differences in the types of feeding guilds and their species compositions within bird communities is informative. Moreover, the responses of functional groups to different forest types must be clarified. For example, having fewer frugivorous birds results in reduced seed dispersal in forests, degrading forest ecosystem function.

Here, we compared bird community assemblages in acacia plantations and fragmented natural forests to evaluate the potential conservation value of each habitat. We then examined feeding guild composition as an indicator of habitat change and fragmentation. Fragmentation scale (i.e., large and small fragments), stands burnt by forest fire, and plantation age were also included in the analyses.

Methods

Study Site

The concession area of the PT. Musi Hutan Persada (PT. MHP) company is located in Muara Enim District, South Sumatra Province, Indonesia (3°00'–4°00' S, 103°00'–104°30' E, Figure 1). The topography is hilly, with an elevation of 60 to 200 m a.s.l. and sedimentary soil. In 2008, when we performed our survey, *A. mangium* was planted on 190,000 ha of the 260,000 ha in the concession area, which is divided into three parts: Wilayah I (region M), Wilayah II (region T), and Wilayah III (not studied). Annual rainfall levels in 2008 were 2,008 mm in region M and 2,849 mm in region T, and mean annual temperature ranges were 25.5°C–28.0°C and 25.4°C–27.9°C, respectively (Shiotani et al. 2009). The dry season lasts from June until September, and the wet season is from November until April. Approximately 95% of the production area is planted with *Acacia mangium*, along with smaller plantations of *Eucalyptus urophylla*, *Pinus merkusii*, *Paraserianthes falcataria*, *Gmelina arborea*, and other species. Trees are harvested on a 6-year rotation and processed as pulp. In 2008, the second-rotation trees were harvested, followed by incremental planting of the third-rotation trees. The plantation was established in 1991 on former “alang-alang” (*Imperata cylindrica*) grasslands, scrublands, and logged areas (Sireger et al. 1998). Based on historical vegetation maps, an estimated one third of the concession area consisted of natural forest in 1985 (WWF-Indonesia 2010), while approximately 80% was secondary forest or forest reserve in 1950 (Hannibal 1950). Thus, the area experienced rapid forest loss between 1950 and 1985, before plantations were established. This deforestation is consistent the estimate of Laumonier (1997), who reported an annual deforestation rate of 1.6% between 1978 and 1985 in southwestern Sumatra, 30 km from our study site. Typical forest transformation in this region occurred first through logging, followed by the allocation of land to transmigration programs or industrial plantations (Laumonier 1997). As a result, South Sumatra province saw abandoned *Imperata* grassland spread over 708,000 ha in the 1990s, more than 6% of the province’s area (Garrity et al. 1997).

Following a new Indonesian law requiring that approximately 10% of the concession area be set aside for conservation (Keputusan Menteri Kehutanan No.70/Kpts-II/95), 25,775 ha, or 9.9% of the total, is now reserved. A network of remnant riparian forests remains mainly along streams and rivers, following a separate regulation aimed at protecting riparian buffers (Peraturan Pemerintah Republik Indonesia No.38/2011 Tentang Sungai), although the conservation area is larger than these riparian forest remnants. However, illegal logging continues throughout the concession area, both in the conservation area and in the riparian forest remnants. Part of the conserved area in both Regions M and T was lost in a large forest fire in 2006, an El-Niño year. Most of the tall trees died, and the forest floor was replaced by dense bush cover by 2008. Some portions of the burnt area have since been planted, either with acacia by the company or dry rice by local residents, and natural invasion of acacia trees has also occurred. Although heavily disturbed and bushy, some parts of the conservation area are in relatively good condition, with more tall trees than in the remnant riparian forests. To assess the conditions of these fragmented forests, we selected three target habitats of varying size: (1) large, logged, forest blocks in the conservation area (LLF); (2) burnt, logged, forest blocks in the conservation area (BLF); and (3) small fragments of remnant, riparian, logged forest (RLF).

We established five survey points in LLF fragments, three in BLF fragments, four in remnant RLF fragments, 12 in 4-year-old acacia plantations (AP4), and eight in 1-year-old acacia plantations (AP1), for a total of 32 points in Regions T and M (Figure 1; Appendix 1). In region T, the two LLF fragments consisted of trees 20–30 m tall interspersed with multistory vegetation, whereas the two BLF fragments were burnt in a forest fire and contained no living tall trees; instead, all sites were covered by dense bush. The remnant RLF fragments consisted of 20-m-tall trees in a narrow strip of forest. The AP4 points consisted of trees 10–20 m tall with less understory (1–10 m) vegetation, and the forest floor was partially covered by ferns or sparse scrub. In contrast, the forest floor of AP1 points had no vegetation, and *A. mangium*

3–5 m in height was the only major plant species. In region M, the three LLF fragments consisted of trees 20 m tall with multistory vegetation, while the one BLF fragment was disturbed and invaded by many acacia trees and ferns. The AP4, AP1, and remnant RLF points in region M were similar to those in region T.

We generated a vegetation map (Figure 1) by combining satellite images from ALOS AVNIR-2 (Advanced Land Observing Satellite “DAICHI”, Advanced Visible and Near Infrared Radiometer type 2) (April 11–May 10, 2008 and October 18, 2010) with the land cover map of Miettinen et al. (2012). The normalized difference vegetation index (NDVI) was calculated from ALOS images, followed by supervised classification using maximum likelihood. Oil palm plantations were masked to exclude them from the classification, because of the potential to confuse them with acacia stands. The coarser land cover map of Miettinen et al. (2010) was included to compensate for masked portions of the ALOS images or those covered by clouds. Of the original categories, “lowland forest” and “lower montane forest” were recategorized as “Woodland/Forest,” “plantation/regrowth” as “Acacia/Regrowth,” and “lowland mosaic” and “lowland open” as “Shrub/Open Land.” Satellite image analysis and mapping were performed with ENVI ver. 4.8 (Exelis VIS, Boulder, CO, USA) and ArcGIS 10 (Esri, Redlands, CA, USA) software. Because of the similar NDVI values among acacia plantations, oil palm plantations, secondary regrowth, and logged-over forest fragments, these land cover types were difficult to differentiate. Therefore, vegetation cover in Figure 1 should be viewed with the caveat that acacia plantations, LLF fragments, BLF fragments, and remnant RLF fragments may be included in both the “Woodland/Forest” and “Acacia/Regrowth” categories.

Bird Survey

We conducted fixed-radius point counts in the wet season of 2007, from 25 October to 9 December, in region T only and in the dry season of 2008, from 11 July to 31 August, in regions T and M. We selected these two seasons to cover all migratory species of the northern and southern winter. Total census times were 2080 min in 2007 and 2920 min in 2008, for a total of 5000 min (Appendix 1). Census locations were all at least 250 m from each other and included two or three sub-points for point-count observation that were themselves at least 50 m apart. Standing at each point, we recorded all species seen in 10 minutes and counted all individuals within a radius of 25 m. We conducted point counts in the morning (06:00–11:30) and evening (14:30–18:30) and ensured that all points were censused in the early morning (06:00–8:00) when birds are most active. Each census continued for a minimum of 15 min. To determine the observation range, we measured a radius of 25 m around each sub-point in advance with a measuring tape and determined whether each observed bird was inside or outside the resulting circle. Bird observations were made by teams of two, each consisting of M. Fujita and a local bird specialist (M. Iqbal, W. Satrio, M. Dwi, and N. Wilson). Bird songs and calls were recorded on an IC-recorder to identify uncertain species later in the lab. We used “Birds of Tropical Asia 3.0” software (Scharringa 2005) to identify unknown bird songs. Individuals that could be identified only to the group level were recorded as “*group name*.sp.”

Vegetation height and cover (%) were estimated by M. Fujita at four different height categories in each habitat type: 0–1, 1–10, 10–20, and 20–30 m. We measured the distance from each point to the nearest conservation area (km) using ArcGIS software. If the nearest conservation area was burnt, we measured the distance to the nearest LLF block instead.

Statistical analyses

We calculated species rarefaction and extrapolation using EstimateS ver. 9.1.0 software (Colwell 2013). As sampling effort differed among habitat types and the species accumulation curve was generally not saturated, comparing species richness using the raw data was difficult. To overcome these problems and to compare species richness more reliably, we extrapolated and calculated estimators. Extrapolation is used to compare smaller with larger samples using statistical sampling models rather than functional curve fitting (Colwell et al. 2012). We calculated up to 200 samples using 95% confidence intervals. Richness estimators (Chao 2, Jackknife 1, and Jackknife 2) were also calculated. In all, 132, 171, 67, 93, and 37 census samples were analyzed for AP1, AP4, BLFs, LLFs, and remnant RLFs, respectively. Individuals that flew over the census point were included in the analyses, but those identified only to the group level were excluded. The Chao 2 richness estimator approaches asymptotic species richness if the sample size is sufficiently large; that is, if the number of unique observations relative to the total number of observations is less than 50% (EstimateS 9.1.0 User's Guide). Our dataset met this condition; therefore, each sample size was large enough to estimate species richness.

Sixteen feeding guilds were classified based on Lambert (1992): R, raptor; TI, terrestrial insectivore; AFGI, arboreal foliage-gleaning insectivore; BGI, bark-gleaning insectivore; SSGI, sallying substrate-gleaning insectivore; SI, sallying insectivore; AI, aerial insectivore; AFGIF, arboreal foliage-gleaning insectivore–frugivore; AF, arboreal frugivore; AFP, arboreal frugivore–predator; TF, terrestrial frugivore; TIF, terrestrial insectivore–frugivore; NI, nectarivore–insectivore; NF, nectarivore–frugivore; NIF, nectarivore–insectivore–frugivore; and MIP, miscellaneous insectivore–piscivore. We also calculated individual-based bird occurrence per census for each feeding guild at each habitat type. Individuals that flew over the census point were excluded from the analyses. Individuals identified only to the group level were included in the analysis by classifying them into the feeding guild in which most species of the same group were categorized. For example, unidentified drongos was classified as SSGI, because most species (e.g., *Dicrurus paradiseus*, *D. remifer*, and *D. aeneus*) belonged to this category. To analyze the deviation of occurrence in each habitat type and feeding guild category, we calculated the squared difference between observed and expected data (deviation, d_{ij}; the same procedure as in the χ^2 test) as follows:

$$d_{ij} = \frac{(O_{ij} - E_{ij})^2}{E_{ij}} \qquad \text{(Equation 1)}$$

where O_{ij} and E_{ij} are the observed and expected values of feeding guild i and habitat type j, respectively.

To analyze differences in community composition among habitat types, we performed an ordination of study sites against density data for each survey year and habitat type, using non-metric multidimensional scaling (NMDS) with the package "vegan" (Oksanen et al. 2010) in R version 3.0.2 (R Development Core Team 2013). NMDS was performed with the Bray–Curtis dissimilarity index and a maximum of 100 iterations to obtain better ordination scores. Correlations between the NMDS axes and environmental variables, such as habitat type (AP1/AP4, BLF/LLF/RLF), vegetation cover (%) at different tree/shrub heights (0–1, 1–10, 10–20, and 20–30 m), and distance to the nearest conservation area (km), were also calculated. We also calculated the species score against each axis, although only abundant species were labeled when several species appeared in close proximity. Individuals that flew over the census point and those identified only to the group level were included in the analyses. To test the degree of similarity among habitat, year, and region, we performed analysis of similarities (ANOSIM), a nonparametric test that statistically evaluates whether there is a significant difference between two or more groups of sampling units (Oksanen 2013). We used the Bray–Curtis dissimilarity index with 999 permutations.

Results

We recorded 103 bird species over 500 censuses (208 in 2007 and 292 in 2008; Appendix 2), accounting for 64.4% of the 160 bird species observed during the study period (Fujita et al. 2010). Observed species richness was higher in AP4 (56 species) and LLF fragments (55 species) than in AP1 (35 species), BLF fragments (37 species), or remnant RLF fragments (37 species) (Table 1). These differences were statistically significant ($P < 0.05$), as the ranges of the respective 95% confidence intervals did not overlap, except slightly between LLF and BLF. The Shannon (H') and Simpson's diversity indices showed that LLF and RLF had higher species diversity than the other habitat types. Species rarefaction and extrapolation up to 200 samples showed that the species accumulation curve of RLFs saturated at approximately 50 species, but those of other forest types did not (Figure 2). One estimator of asymptotic species richness, Chao 2, was highest in LLF fragments (95.2) and AP4s (94.3), followed by BLF fragments

(74.4), AP1s (61.1), and remnant RLFs (50.0) (Table 1). However, the differences among Chao 2 values were not statistically significant, as their respective 95% confidence intervals showed high degrees of overlap. The other estimators, Jackknife 1 and Jackknife 2, yielded similar results, although AP1s and RLFs were reversed in order. Overall bird density (per census abundance) in natural forests (6.2 in RLF; 4.3 in BLF; 3.4 in LLF) was higher than in acacia plantations (3.5 in AP4; 1.9 in AP1) (Table 1). These differences could be even more marked considering the likelihood of underestimating the number of birds in natural forest because of the limited visibility compared with more open acacia plantations. We observed one “vulnerable” species (*Spizaetus nanus*) in BLF in 2007 (BirdLife International 2012; Appendix 2).

We observed species categorized in 14, 16, 13, 14, and 10 of the 16 feeding guilds in AP1s, AP4s, BLFs, LLFs, and remnant RLFs, respectively (Table 2). The dominant feeding guilds across the study region were arboreal foliage gleaning insectivores (AFGI), arboreal foliage gleaning insectivore-frugivores (AFGIF),, nectarivore-insectivore-frugivores (NIF) and sallying insectivores (SI), in descending order of frequency. These four feeding guilds encompassed more than 80% of all individuals observed (Figure 3).

AP4s were home to arboreal foliage gleaning insectivore-frugivores (AFGIF), including medium-sized bulbuls, especially *Pycnonotus goiavier*, and smallnectarivore-insectivore-frugivores (NIF), such as *Dicaeum trigonostigma* and *Nectarinia* *sperata*, but we observed fewer arboreal foliage gleaning insectivores (AFGI) (Table 2). In contrast, AP1s were characterized by more terrestrial insectivore-frugivores (TIF), including *Streptopelia chinensis* and *Geopelia striata*, and terrestrial frugivores (TF), such as *Chalcophaps indica*, but we observed fewer NIFs. Fewer sallying insectivores (SI) and sallying substrate gleaning insectivores (SSGI) were seen in acacia plantations, except for some forest edge species, including *Hemipus hirundinaceus*, *Eurystomus orientalis*, and *Rhipidula javanica*. These results showed that community structure develops according to the age of the acacia trees.

LLF fragments were characterized by a higher incidence of sallying substrate gleaning insectivores (SSGI), such as *Dicrurus* spp. and broadbills, and a lower frequency of arboreal foliage gleaning insectivores (AFGI), such as *Aegithina* spp., *Cacomantis* spp., babblers, woodpeckers, *Pericrocotus* spp., *Phaenicophaeus* spp., and warblers. In contrast, BLF fragments were characterized by arboreal foliage gleaning insectivore-frugivores (AFGIF), such as bulbuls, *Corvus* spp., and *Oriolus* spp.; nectarivore frugivore (NF), such as *Loriculus galgulus*; aerial insectivore (AI), such as *Hirundo rustica*; and only a small number of nectarivore-insectivore-frugivores (NIF), such as sunbirds, flowerpeckers, and *Chloropsis* spp. Finally, remnant RLFs were characterized by AFGIs and terrestrial insectivores (TI), such as *Centropus* spp., some babblers, pittas, and smaller numbers of AFGIFs and AIs.

The NMDS ordination of survey points and subsequent ANOSIM indicated significant differences in community assemblages among habitat types and years but not regions (Figure 4a; Table 3). The similarity in community structure between regions T and M indicated that they can be treated as replicates. Among habitat types, AP1s (0.190) and BLF fragments (0.134) showed higher NMDS 1 values than LLF (−0.214) or remnant RLF fragments (−0.242) (Table 4). We found lower NMDS 2 values for LLF fragments (−0.200) and BLF fragments (−0.170) than for AP1s (0.113) or AP4s (0.074). NMDS 1 correlated positively with shrub cover (vg.0110) and negatively with cover of trees 10–20 m tall (vg.1020). NMDS 2 correlated positively with distance from the nearest conservation area and negatively with understory vegetation cover (vg.0001) and cover of trees 20–30 m tall (vg.2030).

Species correlated with the ordination in logged forests included those found only in LLF fragments (*Pycnonotus cyaniventris*, *Arachnothera flavigaster*, *Surniculus lugubris*, *Alophoixus bres*). Conversely, BLF showed a strong correlation with species that favor open spaces (e.g., *Centropus bengalensis*). Species correlated with the ordination in acacia plantations included those found in AP4s (*Rhipidura javanica*, *Pachycephala grisola*) and AP1s (*Pycnonotus aurigaster*, *Prinia flaviventris*). These results indicated that the NMDS 1 axis represents a gradient from open (positive) to closed-canopy (negative) habitat, while the NMDS 2 axis represents a gradient from cultivated plantation (positive) to natural forest (negative).

Discussion

Estimated species richness was highest in LLF fragments, followed by AP4s and BLF fragments for all three estimators: Chao 2, Jackknife 1, and Jackknife 2 (Table 1). Two estimators ranked AP1s as having higher richness than remnant RLFs. Our results were generally consistent with those of previous studies in Borneo showing that acacia plantations are less diverse than natural forests (Sheldon et al. 2010; Styring et al. 2011).

Our stand-scale survey showed that LLF and RLF fragments have great value in conserving bird diversity. The community compositions of AP1s and AP4s were different from those in logged forests, including LLF, BLF, and RLF fragments (Table 4). The most species-rich LLF fragment was characterized by many forest species that occurred only in such habitats (Figure 4b). Nevertheless, compared to the ordination results of Styring et al. (2011), differences between natural logged forests and acacia plantations were less clear in our study area with respect to species number, feeding guild pattern, and community structure. That is, AP4s showed a community composition that was closer to those of the logged forests in NMDS 1 (Table 4; Figure 4a); AP4s included some species that occurred in logged forests, such as *Pycnonotus goiavier*, *Pycnonotus plumosus*, and *Pycnonotus atriceps* (Figure 4b), which favor open forests and forest edges.

There are several possible reasons for the community similarity and comparably high species richness of AP4s at our study site. First, the logged forests are highly fragmented, and many forest-dependent species have already been lost. Second, the network of RLFs that exists within a matrix of acacia plantations could act as a species source, as many acacia stands are connected or close to RLFs. The high bird density (6.16/census) in RLFs suggests that birds use both RLF fragments and AP4s at the same time. As AP1s are species-poor in the same landscape, AP4s could be used by more bird species than AP1s. More detailed analysis of landscape structure is needed to clarify this possibility.

Although ordination analysis did not detect significant differences in community structure between RLFs and LLFs, there were differences in both total species density and feeding guilds. The feeding guild pattern in LLF fragments showed fewer representatives of arboreal foliage gleaning insectivores (AFGI) and more of arboreal foliage gleaning insectivore-frugivores (AFGIF), whereas RLFs showed the opposite pattern, with fewer AFGIF species, such as *Pycnonotus* spp., and more AFGIs, especially ioras, *Macronous gularis*, *Orthotomus ruficeps*, and *Orthotomus atrogularis*. RLF was also characterized by more terrestrial insectivore (TI) species, especially *Pitta guajana*, *Pellorneum capistratum*, and *Trichastoma rostratum*. These observations indicated that RLFs are more suitable for small insectivores that feed mostly along the forest edge and on the forest floor. LLFs were characterized by more SSGI species, including drongos, *Eurylaimus ochromalus*, and *Rhinomyias umbratilis*. Hornbills were also seen mostly in LLF fragments, which would be suitable for medium-sized fly-catching insectivores and bulbuls.

A significant shift in the bird community occurred in BLFs, probably due to the disappearance of tall trees and the emergence of a dense shrub layer resulting from the large forest fire in 2006 (Figure 4a). BLFs were characterized by more frugivores and fewer insectivores than LLFs, except for *Hirundo rustica*, an aerial insectivore (AI) that was observed once in a flock by chance (Table 2). Despite the disturbance caused by fire, we observed *Spizaetus nanus* in BLF (Appendix 2). Although this species tolerates some disturbance (BirdLife International 2012), whether it will persist in the area is unclear. We also frequently observed hornbills in BLF fragments, although not during the 25-m census. Long-term monitoring of these species will be necessary to fully understand the effects of fire disturbance, as there may be some relaxation time (also known as time lag to extinction) before the disturbed community reaches a new equilibrium (Kuussaari et al. 2009).

Compared to other studies (Danielsen and Heegaard 1995; Thiollay 1995) in the 1990s in lowland forest on Sumatra, many forest species, such as trogons, broadbills, barbets, and babblers, were absent from our study site. In contrast, we found species that favor open habitats or the forest edge, such as warblers, doves, and minivets, which were not recorded in those previous studies. These species were seen in both logged forest and acacia plantations, indicating that the logged forest fragments at our study site do not harbor many species restricted to dense tropical rain forest, the original vegetation on this island. As a result of extensive forest fragmentation across South Sumatra province in the 1970s due to intensive cultivation and logging, this area is now only inhabited by species adapted to fragmentation.

The response of species richness to habitat change was consistent with a previous study by Styring et al. (2011) in Bornean industrial plantations, where more species were observed in natural logged forests and fewer in acacia plantations. The feeding guilds that responded to habitat change were somewhat similar. Birds of the arboreal foliage gleaning insectivores (AFGI), nectarivore-insectivores (NI), and terrestrial insectivores (TI) feeding guilds responded positively, whereas species of the arboreal foliage gleaning insectivore-frugivores (AFGIF), nectarivore-frugivores (NF), raptors (R), sallying substrate gleaning insectivores (SSGI), and terrestrial insectivore-frugivores (TIF) feeding guilds responded negatively to acacia plantations at both sites. Conversely, arboreal frugivores (AF) and arboreal frugivore-predators (AFP) were uncommon at our site, but bark gleaning insectivores (BGI) and nectarivore-insectivore-frugivores (NIF) abundances were relatively high compared to natural Bornean forest. The increase in NIFs at our site was mainly due to the increased population of

Dicaeum trigonostigma, which has also increased in Bornean acacia plantations. Other species that contributed to the increase in acacia plantations at our site were *Anthreptes malacensis*, *Anthreptes simplex*, and *Nectarinia sperata*, the latter of which was not observed at the Bornean site. BGIs showed an even clearer difference between the two sites. None of the species increased in abundance in the Bornean forest, whereas at our site, three of five species did: *Blythipicus rubiginosus*, *Sitta frontalis*, and *Picus mineaceus*; *P. mineaceus* did not occur at the Bornean site. Some of the BGI birds did occur in 7-year old Bornean acacia plantations, but there was still a population decline compared to the natural forest (Styring et al. 2011). These differences in bird response, especially for BGIs, may be related to the availability of ants, termites, and other insects as food sources, along with the forest structure and distance to natural forest patches, but further studies are needed to clarify this point.

The results of the present study suggested that fragmented natural forests harbor richer bird diversity than plantations, a result that is consistent with previous reports (Najera and Simonetti 2009; Edwards et al. 2010). Although fragmented natural forests do not help to conserve primary forest species that would have been present in the past or in other regions, they can harbor other species that are resistant to habitat modification. Fragmented forests play an important role in biodiversity conservation in this region, where large primary forest blocks have been almost entirely lost; this finding mirrors results from other tropical regions (McShea et al. 2009; Turner and Corlett 1996; Chang et al. 2013). A network of small riparian remnant forest fragments could be a complementary habitat for many species, although we found that their species richness was lower than in large fragments. Regardless, if a fragmented forest is burnt, its conservation value decreases. As most of the land in South Sumatra province, and in Sumatra as a whole, is experiencing drastic and ongoing deforestation, there is an urgent need to conserve forest bird species by maintaining the limited remaining natural vegetation. We suggest that (1) even larger conservation areas should be maintained without disturbance, except for some sustainable logging, and (2) wider remnant natural forest strips along rivers and streams should be established.

Acknowledgements

Our great thanks go to Shigeru Shimoda, the former Director of PT. Musi Hutan Persada, for permission to conduct the surveys and for providing accommodation and help during our field work. We also thank Muhammad Iqbal, Wilson Novarino, Dwi Mulyawati, and Satrio Wijamukti for their help in bird identification; Maya Lioe, Bambang Supriadi, Rosyid Gunawan, and Yuli Lestari for their support and discussions during our stay in the field; and all of the field staff for their hard work and patience. We would like to thank Sara Cousins and Michael Lenz for their valuable comments on the manuscript, and Hiromitsu Samejima for his advice on statistical analysis. We are grateful to Betsy Yaap and an anonymous referee for their constructive suggestions to improve our manuscript. This paper was funded by the Research Institute for Sustainable Humanosphere, Kyoto University, and the JSPS Global COE Program "In Search of Sustainable Humanosphere in Asia and Africa" in the Center for Southeast Asian Studies, Kyoto University, Japan.

References

Barlow J, Gardner T, Araujo IS, Avila-Pires TC, Bonaldo B, Costa JE, Esposito MC, Ferreira LV, Hawes J, Hernandez MIM, Hoogmoed MS, Leite RN, Lo-Man-Hung NF, Malcolm JR, Martins MB, Mestre LM, Miranda-Santos R, Nunes-Gutjahr L, Overal WL, Parry L, Peters SL, Ribeiro-Junior M, da Silva MNF, da Silva Motta C, Peres C (2007a) Quantifying the biodiversity value of tropical primary, secondary, and plantation forests. PNAS 104:18555–18560

Barlow J, Mestre LM, Gardner T, Peres C (2007b) The value of primary, secondary and plantation forests for Amazonian birds. Biological Conservation 136:212–231

Beier P, van Drielen M, Kankam BO (2002) Avifaunal collapse in west African forest fragments. Conserv Biol 16:1097−1111

BirdLife International (2012) *Nisaetus nanus*. In: IUCN 2013. IUCN Red List of Threatened Species. Version 2013.2. <www.iucnredlist.org>. Downloaded on 27 March 2014

Chang X, Quan R, Wang L (2013) Bird conservation in extremely small tropical rainforest patches in Southwest China. Biol Conserv 158:188−195

Chung AYC, Eggleton P, Speight MR, Hammond PM, Chey VK (2000) The diversity of beetle assemblages in different habitat types in Sabah, Malaysia. Bull Entomol Res 90:475–496

Colwell RK, Chao A, Gotelli NJ, Lin SY, Mao CX, Chazdon RL, Longino JT. (2012) Models and estimators linking individual-based and sample-based rarefaction, extrapolation, and comparison of assemblages. J Plant Ecol 5:3–21

Colwell RK (2013) EstimateS: Statistical estimation of species richness and shared species from samples. Ver. 9. User's Guide and application published at: http://purl.oclc.org/estimates.

Cossalter C, Pye-Smith C (2003) Fast-wood forestry myths and realities, CIFOR, Bogor.

Danielsen F, Heegaard M (1995) The birds of Bukit Tigapuluh, southern Riau, Sumatra. KUKILA 7:99–120

Edwards DP, Hodgson JA, Hamer KC, Mitchell SL, Ahmad AH, Cornell SJ, Wilcove DS (2010) Wildlife-friendly oil palm plantations fail to protect biodiversity effectively. Conserv Lett 3:236−242

FAO (2010) Global Forest Resources Assessment 2010 Main Report. FAO.

Fujita MS, Yoshimura T, Iqbal M, Wijamukti S, Mulyawati D, Novarino W, Lestari Y, Supriadi B, Gunawan R, Prawiradilaga DM (2010) Inventory of birds in *Acacia* plantation in PT. Musi Hutan Persada, Indonesia. Kyoto Working Papers Area Stud 110:1–49

Gardner T, Barlow J, Chazdon R, Ewers RM, Harvey C, Peres C, Sodhi NS (2009) Prospects for tropical forest biodiversity in a human-modified world. Ecol Lett 12:561–582

Garrity D, Soekardi M, Noordwijk MV, De La Cruz R, Pathak P, Gunasena HP, So NV, Huijun G, Majid NM (1997) The *Imperata* grasslands of tropical Asia: area, distribution, and typology. Agroforest Syst 36:3–29

Hannibal LW (1950) Vegetation map of Indonesia. Planning Department of the Forest Service, Jakarta

Hawes J, Barlow J, Gardner T, Peres C (2008) The value of forest strips for understorey birds in an Amazonian plantation landscape. Biol Conserv 141:2262–2278

Hill JK, Gray M, Khen CV, Benedick S, Tawatao N, Hamer KC (2011) Ecological impacts of tropical forest fragmentation: how consistent are patterns in species richness and nestedness? Philos Trans Roy Soc B 366:3265–3276

Kuussaari M, Bommarco R, Heikkinen RK, Helm A, Krauss J, Lindborg R, Öckinger E, Pärtel M, Pino J, Rodà F, Stefanescu C, Teder T, Zobel M, Steffan-Dewenter I (2009) Extinction debt: a challenge for biodiversity conservation. Trends Ecol Evol 24:564–571

Lambert FR (1992) The consequences of selective logging for Bornean lowland forest birds. Philos Trans Roy Soc B 335:443–457

Laumonier Y (1997) *The Vegetation and Physiography of Sumatra* (Vol. 22). Kluwer Academic Publishers, Dordrecht.

Laurance WF (2007) Forest destruction in tropical Asia. Curr Sci 93:1544−1550

Laurance SG, Laurance WF (1999) Tropical wildlife corridors: use of linear rainforest remnants by arboreal mammals. Biol Conserv 91:231−239

Lindenmayer DB, Cunningham RB, Donnelly CF, Nix H, Lindenmayer BD (2002) Effects of forest fragmentation on bird assemblages in a novel landscape context. Ecol Monogr 72:1–18

Marquis RJ, Whelan CJ (1994) Insectivorous birds increase growth of white oak through consumption of leaf-chewing insects. Ecology 75:2007−2014

McShea WJ, Stewart C, Peterson L, Erb P, Stuebing R, Giman B (2009) The importance of secondary forest blocks for terrestrial mammals within an Acacia/secondary forest matrix in Sarawak, Malaysia. Biol Conserv 142:3108−3119

Miettinen Y, Chenghua S, Juan TW, Chin SC (2012) 2010 land cover map of insular Southeast Asia in 250-m spatial resolution. Remote Sens Lett 3:11−20

Millennium Ecosystem Assessment (2005) Ecosystems and human well-being: Synthesis. Island Press, Washington, DC

Myers N, Mittermeier RA, Mittermeier CG, Fonseca GAB, Kent J (2000) Biodiversity hotspots for conservation priorities. Nature 403:853–859

Najera A, Simonetti JA (2009) Enhancing avifauna in commercial plantations. Conserv Biol 24:319−324

Nasi R, Koponen P, Poulsen JG, Buitenzorgy M, Rusmantoro W (2008) Impact of landscape and corridor design on primates in a large-scale industrial tropical plantation landscape. Biodiv Conserv 17:1105–1126

O’Connell TJ, Jackson LE, Brooks RP (2000) Bird guilds as indicators of ecological condition in the central Appalachians. Ecol Appl 10:1706–1721

Oksanen JF, Blanchet G, Kindt R, Legendre P, Minchin PR, O'Hara RB, Simpson GL, Solymos P, Stevens MHM, Wagner H (2010) vegan: Community Ecology Package. R package ver. 1.17-4. http://CRAN.R-project.org/package=vegan

Oksanen JF (2013) Package “vegan”. http://cran.r-project.org/web/packages/vegan/vegan.pdf

R Development Core Team (2013) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. http://www.R-project.org

Ricketts TH, Daily GC, Ehrlich PR, Michener CD (2004) Economic value of tropical forest to coffee production. PNAS 101:12579–12582

Scharringa J (2005) Birds of Tropical Asia 3.0. Bird Songs International. http://www.birdsongs.com/TropicalAsia/main.htm

Sheldon FH, Styring A, Hosner P (2010) Bird species richness in a Bornean exotic tree plantation: A long-term perspective. Biol Conserv 143:399–407

Shiotani et al. (2009) Database for meteorological observation in acacia plantation in Indonesia (*in Japanese*). http://database.rish.kyoto-u.ac.jp/arch/acacia/index.html (accessed on 24 August 2012)

Sireger STH, Hardiyanto EB, Gales K (1998) *Acacia mangium* plantations in PT Musi Hutan Persada, South Sumatera, Indonesia. In: Nambiar EKS, Cossalter C, Tiarks A (eds) Workshop Proceedings: Site Management and Productivity in Tropical Plantation Forests. CIFOR, Bogor, pp 39−44

Sodhi NS, Brook B (2006) Southeast Asian biodiversity in crisis. Cambridge University Press, Cambridge, UK

Sodhi NS, Koh LP, Brook BW, Ng PKL (2004) Southeast Asian biodiversity: an impending disaster. Evolution 19:654−660

Sodhi NS, Lee TM, Koh LP, Brook BW (2009) A meta-analysis of the impact of anthropogenic forest disturbance on Southeast Asia’s biotas. Biotropica 41:103–109

Struebig MJ, Kingston T, Zubaid A, Mohd-Adnan A, Rossiter SJ (2008) Conservation value of forest fragments to Palaeotropical bats. Biol Conserv 141:2112−2126

Styring AR, Ragai R, Unggang J, Stuebing R, Hosner P, Sheldon FH (2011) Bird community assembly in Bornean industrial tree plantations: Effects of forest age and structure. Forest Ecol Manag 261:531–544

Thiollay JM (1995) The Role of traditional agroforests in the conservation of rain forest bird diversity in Sumatra. Conserv Biol 9:335–353

Tsukamoto J, Sabang J (2005) Soil macro-fauna in an *Acacia mangium* plantation in comparison to that in a primary mixed dipterocarp forest in the lowlands of. Pedobiologia 49:69−80

Turner IM, Corlett RT (1996) The conservation value of small, isolated fragments of lowland tropical rain forest. Trends Ecol Evol 11:330−333

Wunderle Jr JM (1997) The role of animal seed dispersal in accelerating native forest regeneration on degraded tropical lands. Science 99:223–235

WWF-Indonesia (2010) Sumatra’s Forests, their Wildlife and the Climate -Windows in Time: 1985, 1990, 2000 and 2009. WWF, Jakarta

Yokota Y, Inoue M (1996) Transmigrasi-HTI in Indonesia: Focusing on a case study in Southern Sumatra. Bull Tokyo Univ Forests 95:209−246 (in Japanese with English summary)

Figure Legends

Figure 1. Study area of PT. Musi Hutan Persada, South Sumatra, Indonesia.

Thirty-two study points were located in Regions M and T: eight of 1-year-old acacia plantation (white triangles, AP1); 12 of 4-year-old acacia plantation (black triangle, AP4); three of burnt logged forest (white circles, BLF); five of large logged forest (black circles, LLF); and four of remnant logged forest (transparent crosses, RLF).

Figure 2. Estimated species accumulation of each habitat type from 10-minutes censuses.

Solid lines show rarefaction with reference samples, while broken lines show extrapolation. Abbreviations for habitat types are: AP1, 1-year-old acacia plantation; AP4, 4-year-old acacia plantation; BLF, burnt logged forest; LLF, large logged forest; RLF, remnant logged forest. EstimateS ver. 9.1.0 (Colwell 2013) was used to compute rarefaction and extrapolation.

Figure 3. Individual-based percentages of bird feeding guilds by habitat.

Observed individuals inside a 25 m radius per 10-minute census were averaged for both 2007 and 2008, except for remnant logged forest, where only data from 2008 are shown. Abbreviations for habitat types are: AP1, 1-year-old acacia plantation; AP4, 4-year-old acacia plantation; BLF, burnt logged forest; LLF, large logged forest; RLF, remnant logged forest. Abbreviations for feeding guilds shown here are: R, raptor; TI, terrestrial insectivore; AFGI, arboreal foliage-gleaning insectivore; BGI, bark-gleaning insectivore; SSGI, sallying substrate-gleaning insectivore; SI, sallying insectivore; AI, aerial insectivore; AFGIF, arboreal foliage-gleaning insectivore–frugivore; AF, arboreal frugivore; AFP, arboreal frugivore–predator; TF, terrestrial frugivore; TIF, terrestrial insectivore–frugivore; NI, nectarivore–insectivore; NF, nectarivore–frugivore; NIF, nectarivore–insectivore–frugivore; MIP, miscellaneous insectivore–piscivore. Classification of feeding guilds is based on Lambert (1992).

Figure 4. NMDS ordination of study points (a) and species (b) by bird community composition.

(a) Each point corresponds to a study point in a season. Arrows and text in the plot indicate correlations between community structure and environmental variables such as habitat type, vegetation, and distance to the nearest conservation area. Abbreviations in plots are as follows: vg.0001, vegetation cover (%) less than 1 m in height; vg.0110, vegetation cover (%) 1–10 m in height; vg.1020, vegetation cover (%) 10–20 m in height; vg.2030, vegetation cover (%) 20–30 m in height. Abbreviations in legends are as follows: T8, region T in 2008 (triangle); M8, region

M in 2008 (circle); T7, region T in 2007 (square); AP1, 1-year-old acacia plantation (blank); AP4, 4-year-old acacia plantation (light gray with black outline); LLF, large logged forest (black); BLF, burnt logged forest (light gray); RLF, remnant logged forest (dark gray). (b) In the species plot, only abundant species are labeled where several species are clustered. Abbreviations for species name are: Aracflav, *Arachnothera flavigaster*; Araclong, *Arachnothera longirostra*; Artaleuc, *Artamus leucorynchus*; Copssaul, *Copsychus saularis*; Geopstri, *Geopelia striata*; Gracreli, *Gracula religiosa*; Hirurust, *Hirundo rustica*; Hypoazur, *Hypothymis azurea*; Lanitigr, *Lanius tigrinus*; Meroviri, *Merops viridis*; Orthatro, *Orthotomus atrogularis*; Orthrufi, *Orthotomus ruficeps*; Pachgris, *Pachycephala grisola*; Pittguaj, *Pitta guajana*; Phaecurv, *Phaenicophaeus curvirostris*; Pycnatri, *Pycnonotus atriceps*; Pycngoia, *Pycnonotus goiavier*; Pycnmela, *Pycnonotus melanicterus*; Pycnplum, *Pycnonotus plumosus*; Pycnsp, Pycnonotus sp.; Rhipjava, *Rhipidula javanica*; Strechin, *Streptopelia chinensis*; sunbsp, sunbird sp.; Tephgula, *Tephrodornis gularis*.

Table 1. Summary of bird diversity in each habitat type.

	Habitat type				
	AP1	AP4	BLF	LLF	RLF
Species richness and diversity					
Species richness S (est.)	35	56	37	55	37
% Species / total species	34.0	54.4	35.9	53.4	35.9
S (est.) 95% CI lower bound	26.68	46.03	27.99	45.18	30.51
S (est.) 95% CI upper bound	43.32	65.97	46.01	64.82	43.49
Shannon diversity index (*H'*)	2.98	3.09	2.71	3.32	3.11
Simpson diversity index	13.08	11.85	8.76	15.90	16.23
Estimators					
Chao 2 mean	61.05	94.27	74.43	95.16	49.97
Chao 2 95% CI lower bound	42.39	68.98	49.05	70.71	40.96
Chao 2 95% CI upper bound	126.86	168.83	153.31	157.68	79.5
Jackknife 1 mean	49.89	77.87	56.7	83.69	52.57
Jackknife 2 mean	61.73	94.7	72.28	103.35	60.35
General information					
Censuses	132	171	67	93	37
Observed total individuals (unidentified)[3]	263 (19)	607 (55)	323 (17)	316 (23)	228 (6)
Uniques[2] mean	15	22	20	29	16
Per-census abundance (indiv./census)	1.91	3.50	4.90	3.42	6.16

[1] Abbreviations for habitat types are: AP1, 1-year-old acacia plantation; AP4, 4-year-old acacia plantation; BLF, burnt logged forest; LLF, large logged forest; RLF, remnant logged forest.

[2] Uniques are species that occurred in only one sample.

[3] Numbers of unidentified individuals out of total individuals are shown in parentheses.

Table 2. Differences (*d*) of observed from expected number of individuals for each feeding guild and habitat type. A negative sign (−) before the value indicates that the observed number was below expectation.

	Habitat type[1]				
Feeding guild[2]	AP1	AP4	BLF	LLF	RLF
AF	0.031	0.007	0.000	−0.005	−0.012
AFGI	0.001	−0.004	−0.216***	−0.278***	0.695***
AFGIF	−0.005	0.000	0.183**	0.082	−0.318***
AFP	−0.002	0.009	−0.005	0.016	−0.006
AI	−0.015	−0.058	1.179***	−0.119**	−0.214***
BGI	0.001	0.004	−0.004	0.020	−0.013
MIP	0.007	0.004	−0.007	0.008	−0.008
NF	0.007	−0.020	0.214***	−0.017	−0.064
NI	−0.000	0.015	−0.050	0.005	0.004
NIF	−0.005	0.033	−0.179**	0.008	0.044
R	0.000	−0.000	−0.000	0.045	−0.019
SI	0.013	0.009	−0.028	0.009	−0.003
SSGI	−0.035	−0.000	−0.051	0.445***	−0.035
TF	0.127**	0.000	−0.012	0.001	−0.015
TI	−0.033	−0.023	−0.021	−0.040	0.244***
TIF	0.157**	−0.035	−0.029	−0.050	0.057

*** $d > 0.200$; ** $d > 0.100$; * $d > 0.050$

[1] Abbreviations for habitat types are: AP1, 1-year-old acacia plantation; AP4, 4-year-old acacia plantation; BLF, burnt logged forest; LLF, large logged forest; RLF, remnant logged forest.

[2] Abbreviations for feeding guilds shown here are: R, raptor; TI, terrestrial insectivore; AFGI, arboreal foliage-gleaning insectivore; BGI, bark-gleaning insectivore; SSGI, sallying

substrate-gleaning insectivore; SI, sallying insectivore; AI, aerial insectivore; AFGIF, arboreal foliage-gleaning insectivore–frugivore; AF, arboreal frugivore; AFP, arboreal frugivore–predator; TF, terrestrial frugivore; TIF, terrestrial insectivore–frugivore; NI, nectarivore–insectivore; NF, nectarivore–frugivore; NIF, nectarivore–insectivore–frugivore; MIP, miscellaneous insectivore–piscivore.

Table 3. Results of the analysis of similarities (ANOSIM).

	Statistic *R*	*P* value
habitat (5 groups)	0.3260	0.001
year (2 groups)	0.1852	0.01
region (2 groups)	0.0795	0.067

Table 4. Mean value of NMDS 1 and 2 in each habitat type and year.

	NMDS1[3]			NMDS2		
Habitat type[2]	Mean	2007	2008	Mean	2007	2008
AP1	0.190	0.303	0.133	0.113	−0.027	0.183
AP4	−0.030	0.068	−0.063	0.074	0.027	0.089
BLF	0.134	0.157	0.118	−0.170	−0.362	−0.042
LLF	−0.214	−0.048	−0.281	−0.200	−0.234	−0.186
RLF	−0.242	n.d.[1]	−0.242	−0.070	n.d.[1]	−0.070

[1] No data available.

[2] Abbreviations for habitat types are: AP1, 1-year-old acacia plantation; AP4, 4-year-old acacia plantation; BLF, burnt logged forest; LLF, large logged forest; RLF, remnant logged forest.

[3] NMDS refer to non-metric multidimensional scaling.

Figures

Figure 1

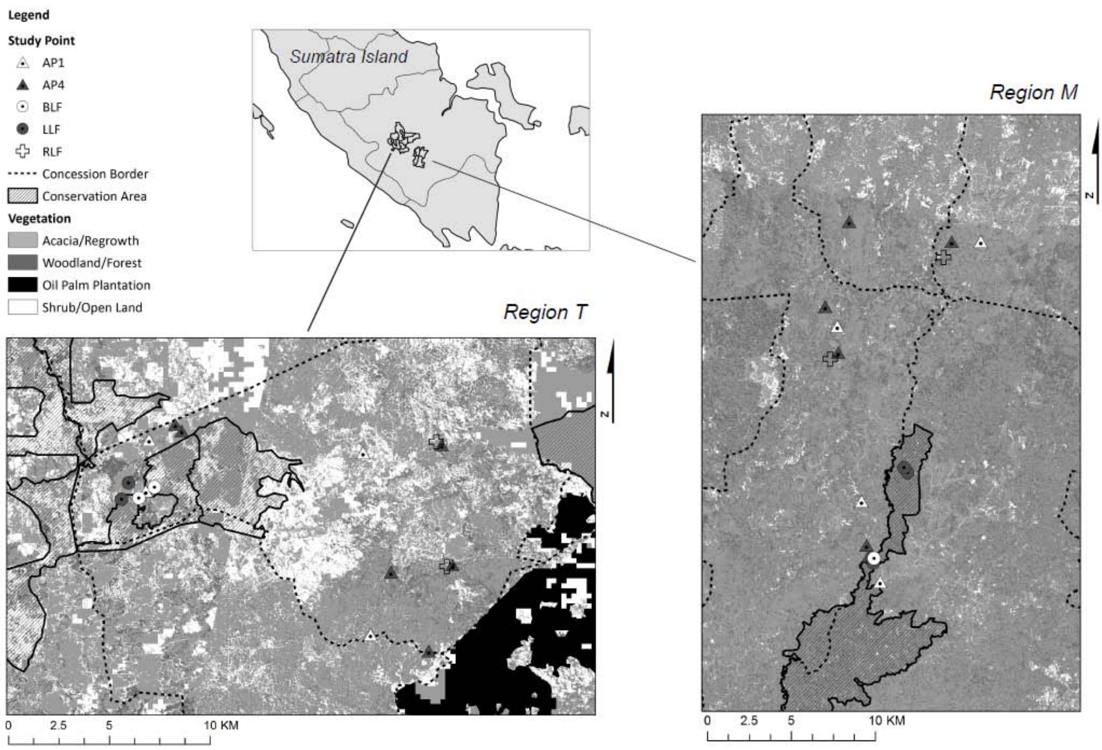


Figure 2

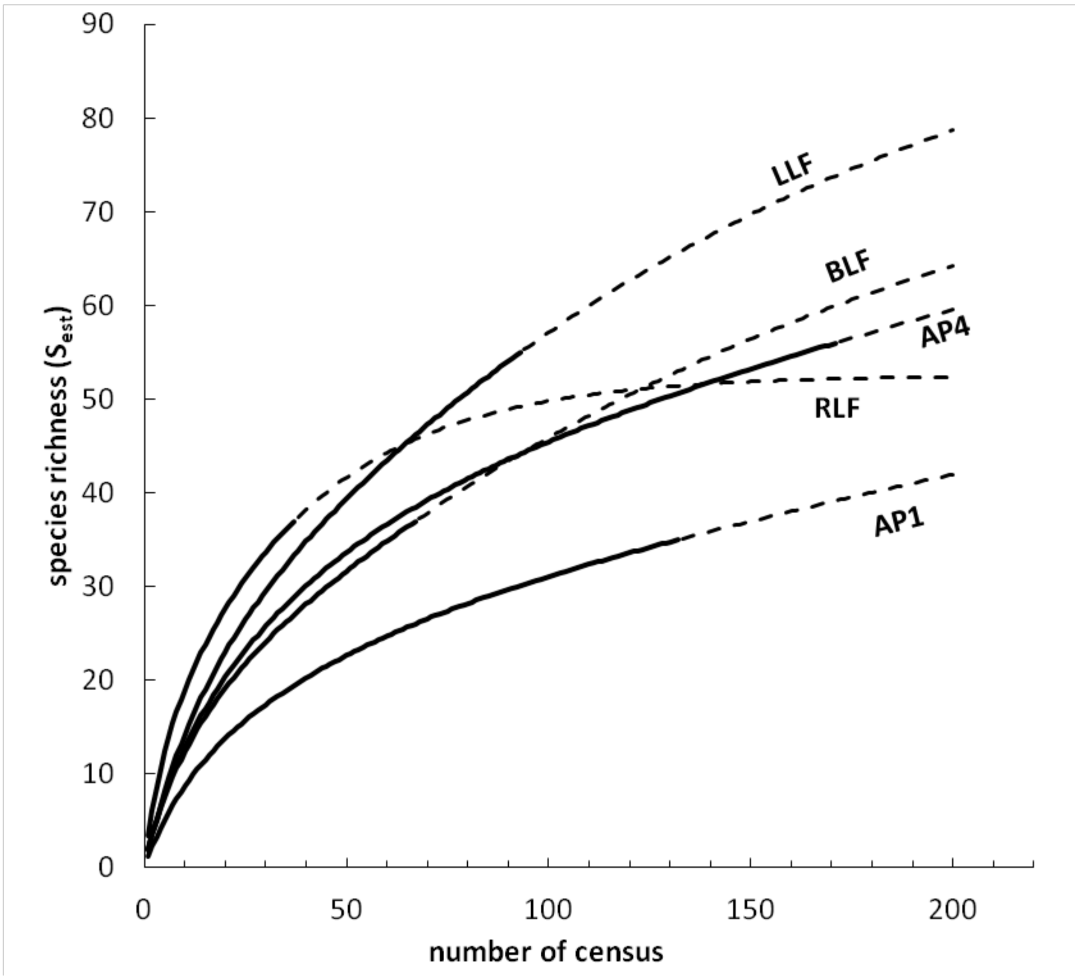

species richness (S_{est})
number of census
90
80
70
60
50
40
30
20
10
0
0
50
100
150
200
LLF
BLF
AP4
RLF
AP1

Figure 3

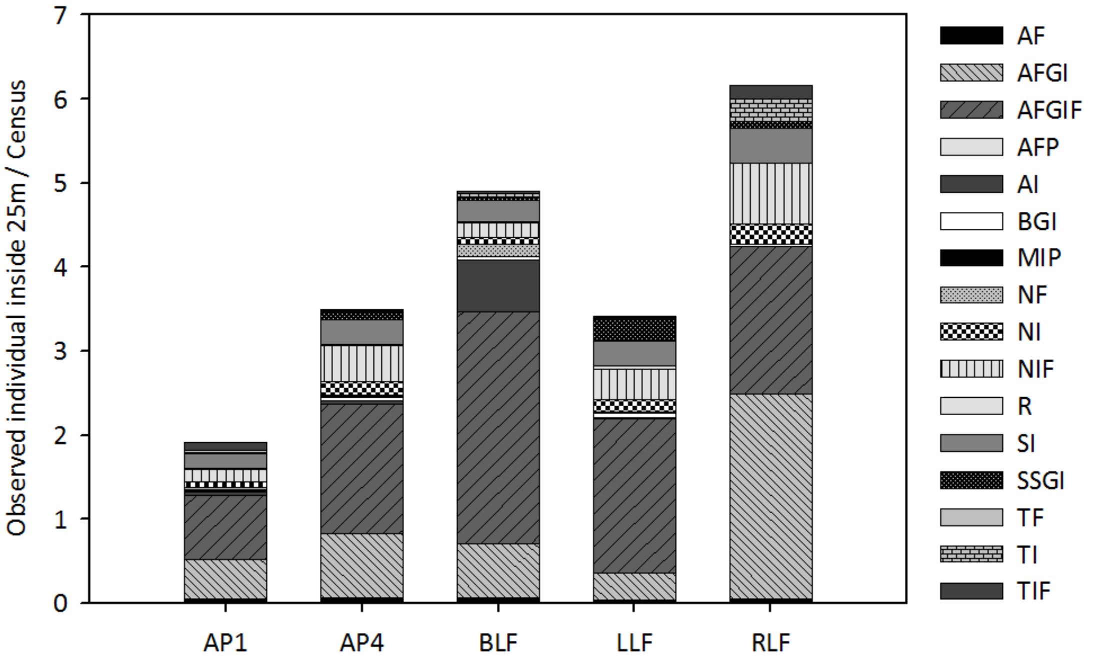
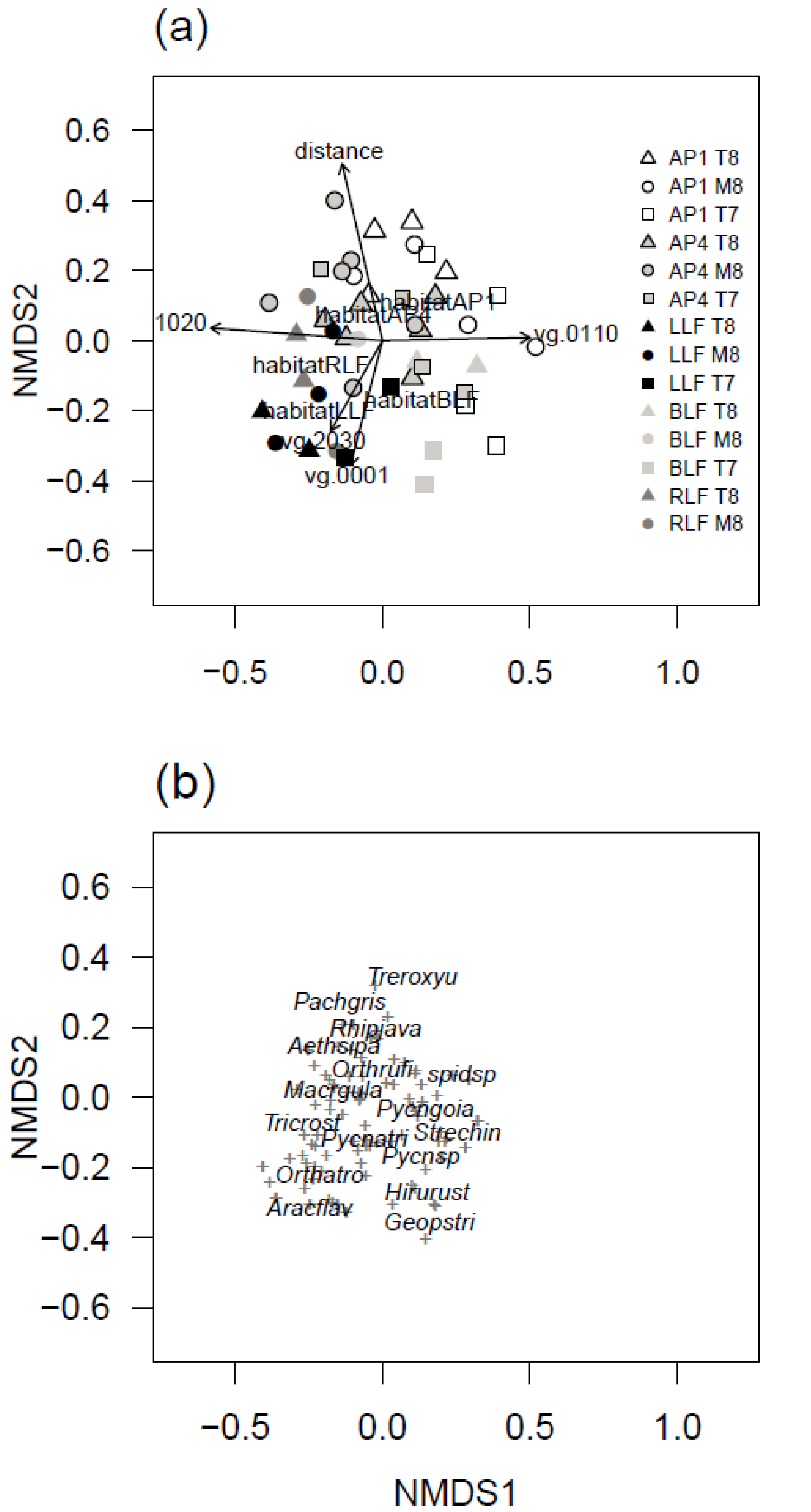

Observed individual inside 25m / Census
0
1
2
3
4
5
6
7
AP1
AP4
BLF
LLF
RLF
AF
AFGI
AFGIF
AFP
AI
BGI
MIP
NF
NI
NIF
R
SI
SSGI
TF
TI
TIF

Figure 4



(a)
NMDS2
0.6
0.4
0.2
0.0
−0.2
−0.4
−0.6
−0.5
0.0
0.5
1.0
distance
1020
vg.0110
habitatAP1
habitatAP4
habitatRLF
habitatLLF
habitatBLF
vg.2030
vg.0001
AP1 T8
AP1 M8
AP1 T7
AP4 T8
AP4 M8
AP4 T7
LLF T8
LLF M8
LLF T7
BLF T8
BLF M8
BLF T7
RLF T8
RLF M8
(b)
NMDS2
0.6
0.4
0.2
0.0
−0.2
−0.4
−0.6
Treroxyu
Pachgris
Rhipiava
Aethsipa
Orthrufi
spidsp
Macrgula
Pycngoia
Tricrost
Pycnatri
Strechin
Pycnsp
Orthatro
Hirurust
Aracflav
Geopstri
−0.5
0.0
0.5
1.0
NMDS1

Appendix 1. Descriptions of study points.

Habitat type	Point name	Region	Latitude	Longitude	Vegetation cover (%) in each tree height (m)				Distance to nearest conservation area (km)	Census time	
					20–30 m	10–20 m	1–10 m	< 1 m		2007	2008
1-year-old acacia plantation (AP1)	A1-01	M	S3° 53' 50.6"	E103° 56' 36.1"	0	0	100	3	0.31		9
	A1-02	M	S3° 42' 53.5"	E103° 59' 50.0"	0	0	100	0	11.551		9
	A1-03	M	S3° 51' 10.8"	E103° 56' 5.2"	0	0	100	0	1.517		9
	A1-04	M	S3° 45' 37.8"	E103° 55' 13.3"	0	0	100	3	7.391		9
	A1-05	T	S3° 24' 33.3"	E103° 31' 42.0"	0	0	100	10	0.256	15	9
	A1-06	T	S3° 28' 24.1"	E103° 37' 42.9"	0	0	100	3	9.14	15	9
	A1-07	T	S3° 23' 11.0"	E103° 31' 47.1"	0	0	100	1	0.783	12	9
	A1-08	T	S3° 22' 9.1"	E103° 37' 30.7"	0	0	100	3	5.715	18	9
4-year-old acacia plantation (AP4)	A4-01	M	S3° 53' 4.1"	E103° 56' 14.0"	0	80	70	15	0.281		9
	A4-02	M	S3° 42' 15.2"	E103° 55' 35.7"	0	80	5	5	12.636		9
	A4-03	M	S3° 42' 54.3"	E103° 58' 54.2"	0	70	5	10	11.056		9
	A4-04	M	S3° 52' 39.6"	E103° 56' 9.8"	0	80	5	100	0.719		9
	A4-05	M	S3° 44' 59.6"	E103° 54' 49.8"	0	80	30	80	8.763		9
	A4-06	M	S3° 46' 28.6"	E103° 55' 14.8"	0	70	30	5	6.252		9
	A4-07	T	S3° 22' 56.6"	E103° 32' 39.6"	0	40	60	100	0.4	19	9

	A4-08	T	S3° 28' 50.7"	E103° 39' 16.5"	0	80	10	80	10.764	15	9
	A4-09	T	S3° 26' 32.0"	E103° 39' 54.4"	0	80	20	60	6.635		9
	A4-10	T	S3° 22' 46.2"	E103° 32' 28.9"	0	30	50	100	0.837	14	9
	A4-11	T	S3° 26' 43.5"	E103° 38' 15.6"	0	70	10	80	8.318		9
	A4-12	T	S3° 23' 18.9"	E103° 39' 37.5"	0	80	50	80	4.761	15	9
Burnt logged forest (BLF)	BF-01	M	S3° 53' 5.5"	E103° 56' 24.3"	0	40	80	5	0.114		9
	BF-02	T	S3° 24' 26.9"	E103° 31' 55.6"	5	20	80	90	0.514	20	11
	BF-03	T	S3° 24' 44.4"	E103° 31' 30.4"	0	1	80	80	0.159	18	9
Large logged forest (LLF)	LF-01	M	S3° 53' 15.4"	E103° 56' 21.3"	0	80	40	5	0		9
	LF-02	M	S3° 50' 2.0"	E103° 57' 28.9"	0	60	10	30	0		9
	LF-03	M	S3° 50' 10.2"	E103° 57' 22.0"	0	90	10	20	0		10
	LF-04	T	S3° 24' 20.3"	E103° 31' 14.2"	10	40	70	95	0	23	9
	LF-05	T	S3° 24' 48.5"	E103° 31' 7.8"	60	40	40	80	0	24	9
Remnant logged forest (RLF)	RF-01	M	S3° 43' 24.4"	E103° 58' 38.3"	0	60	70	50	10.054		9
	RF-02	M	S3° 46' 40.7"	E103° 54' 59.5"	0	70	70	40	6.283		10
	RF-03	T	S3° 26' 34.6"	E103° 39' 45.6"	0	80	40	20	6.879		9
	RF-04	T	S3° 23' 14.1"	E103° 39' 28.8"	0	5	100	20	4.75		9

653

Appendix 2. List of species occurrence per census in each habitat type.

Scientific name	Habitat type	AP1			AP4			BLF			LLF			RLF	
	Region	M	T	T	M	T	T	M	T	T	M	T	T	M	T
	Year	8	7	8	8	7	8	8	7	8	8	7	8	8	8
	Census	36	60	36	54	63	54	9	38	20	28	47	18	19	18
	Feeding guild														
Treron oxyura	AF	0	0	0.03	0	0	0	0	0	0	0	0	0	0	0
Treron curvirostra	AF	0	0	0	0	0	0	0	0	0.05	0	0	0	0	0
Treron vernans	AF	0	0.07	0	0	0.08	0.09	0	0.05	0	0	0	0	0	0.11
Irena puella	AF	0	0	0	0	0	0	0	0.03	0	0	0	0.11	0	0
Gracula religiosa	AF	0	0	0.03	0	0	0	0	0	0	0	0	0	0	0
Cacomantis sonneratii	AFGI	0	0	0	0	0	0	0	0	0	0	0.02	0	0	0
Cacomantis merulinus	AFGI	0	0	0	0	0.02	0	0	0.03	0	0	0	0	0.05	0
Phaenicophaeus diardi	AFGI	0	0	0.03	0	0	0	0	0	0	0	0	0	0.05	0
Phaenicophaeus chlorophaeus	AFGI	0	0	0	0	0	0	0	0	0	0	0	0.11	0	0
Phaenicophaeus curvirostris	AFGI	0	0	0	0	0	0	0	0	0	0	0	0	0	0.17
Sasia abnormis	AFGI	0	0	0	0	0	0.02	0	0	0	0.04	0	0	0	0
Dendrocopos canicapillus	AFGI	0	0	0	0	0	0	0	0	0.05	0	0	0	0	0
Dendrocopos moluccensis	AFGI	0	0	0	0.02	0.02	0.02	0	0	0	0	0	0	0	0
Tephrodornis gularis	AFGI	0	0	0	0	0	0	0	0	0	0	0.02	0.06	0	0
Coracina fimbriata	AFGI	0	0	0	0	0	0	0	0.03	0	0.07	0	0	0.05	0
Pericrocotus igneus	AFGI	0	0	0	0	0	0	0	0.11	0.2	0	0.04	0	0.11	0
Pericrocotus solaris	AFGI	0	0	0	0	0	0	0.11	0	0	0	0	0	0	0
Aegithina viridissima	AFGI	0	0	0	0.02	0	0	0	0	0	0	0	0	0.16	0.11
Aegithina tiphia	AFGI	0	0	0.03	0.06	0.06	0.04	0.11	0	0	0	0	0.06	0.47	0
Malacocincla sepiarium	AFGI	0.06	0	0	0	0	0	0	0	0	0	0	0	0	0
Stachyris erythroptera	AFGI	0	0	0	0	0	0.06	0	0	0	0	0	0	0	0
Macronous gularis	AFGI	0.17	0	0.08	0.15	0	0.15	0	0	0.2	0	0	0.06	0	1.33
Copsychus saularis	AFGI	0.06	0.03	0	0.06	0.05	0	0	0	0	0	0.02	0	0	0
Phylloscopus inornatus	AFGI	0	0	0	0	0	0	0	0	0	0.04	0	0	0	0
Orthotomus atrogularis	AFGI	0	0	0	0	0	0	0.22	0	0	0.14	0	0	0.58	0.22
Orthotomus ruficeps	AFGI	0.17	0.02	0.33	0.39	0.21	0.43	0.22	0	0.2	0	0.02	0.22	0.74	0.11
Orthotomus sericeus	AFGI	0.08	0	0.08	0.09	0	0.17	0	0	0	0.04	0	0.22	0.11	0.44
Prinia flaviventris	AFGI	0.25	0.02	0.14	0	0.08	0.07	0	0.03	0.35	0.04	0	0	0	0
Prinia familiaris	AFGI	0	0.05	0	0	0.02	0	0	0.11	0.1	0	0	0	0	0
Pachycephala grisola	AFGI	0	0	0	0.07	0	0	0	0	0	0	0	0	0	0
Calorhamphus fuliginosus	AFGIF	0	0	0	0	0	0	0	0	0.1	0	0	0	0	0
Pycnonotus atriceps	AFGIF	0.14	0.13	0	0.35	0.08	0.13	1	0.47	0.55	1.25	0.3	0.28	1.21	0.44
Pycnonotus melanicterus	AFGIF	0.03	0	0.14	0.13	0	0.04	0.44	0	0.25	0.39	0.04	0.28	0.32	0.44
Pycnonotus cyaniventris	AFGIF	0	0	0	0	0	0	0	0	0	0	0	0.17	0	0
Pycnonotus aurigaster	AFGIF	0.33	0.03	0.22	0.13	0.21	0.24	0.11	0	0.65	0	0	0	0	0
Pycnonotus goiavier	AFGIF	0.53	0.23	0.42	0.11	1.16	0.8	0.44	0.97	1.8	0.14	0.26	0	0	0.06
Pycnonotus plumosus	AFGIF	0.22	0	0.19	0.13	0.02	0.11	1	0	0.3	0.43	0.06	0.11	0.16	0.39
Pycnonotus simplex	AFGIF	0	0	0	0.02	0	0.02	0	0	0	0.04	0	0.5	0	0.06
Pycnonotus brunneus	AFGIF	0	0	0	0.02	0.11	0.28	0	0.29	0.25	0.29	0.28	0.17	0	0.22
Pycnonotus erythropthalmos	AFGIF	0	0	0	0	0.02	0	0	0	0	0.14	0.04	0.17	0	0.11
Alophoixus bres	AFGIF	0	0	0	0	0	0	0	0	0	0.07	0	0.11	0	0
Tricholestes criniger	AFGIF	0	0	0	0.02	0	0	0	0	0	0.04	0	0.17	0	0.11
Ixos malaccensis	AFGIF	0	0	0	0	0	0	0	0	0	0.11	0	0	0	0
Corvus enca	AFGIF	0	0	0	0	0.02	0	0	0	0	0	0.04	0	0	0
Anthracoceros malayanus	AFP	0	0	0	0.04	0	0	0	0	0	0	0	0	0	0
Buceros rhinoceros	AFP	0	0	0	0	0	0	0	0	0	0	0	0.06	0	0
Hemiprocne longipennis	AI	0	0	0	0	0.05	0.06	0	0	0	0	0	0	0	0
Hirundo rustica	AI	0	0	0.14	0	0	0	0	1.24	0	0	0	0	0	0
Celeus brachyurus	BGI	0	0	0	0	0	0	0	0	0	0.07	0	0	0	0
Picus mineaceus	BGI	0	0	0	0	0	0.02	0	0	0	0	0	0	0	0
Blythipicus rubiginosus	BGI	0	0	0	0.02	0	0	0	0	0	0	0	0	0	0
Reinwardtipicus validus	BGI	0	0	0	0	0	0	0	0	0	0	0	0.11	0	0
Sitta frontalis	BGI	0.08	0	0	0.04	0	0	0	0	0	0	0	0	0	0

Alcedo meninting	MIP	0	0	0	0	0	0.02	0	0	0	0	0	0	0	0
Lacedo pulchella	MIP	0	0	0	0	0	0	0	0	0	0	0	0.06	0	0
Loriculus galgulus	NF	0	0.05	0.03	0	0	0.04	0	0.11	0.3	0	0	0.06	0	0
Anthreptes rhodolaema	NI	0	0	0	0	0	0	0	0	0.1	0	0	0	0	0
Anthreptes singalensis	NI	0	0	0	0	0	0	0	0.03	0	0	0	0	0	0.06
Nectarinia jugularis	NI	0.03	0	0	0.07	0.03	0	0	0	0	0.07	0	0	0	0
Aethopyga siparaja	NI	0	0	0	0.09	0.03	0.09	0	0	0	0	0	0	0.26	0.06
Arachnothera longirostra	NI	0	0	0	0.07	0	0	0	0	0	0.18	0	0.06	0	0.06
Arachnothera crassirostris	NI	0	0	0	0	0	0	0	0.03	0	0.07	0	0	0	0
Arachnothera chrysogenys	NI	0	0	0	0	0	0	0	0	0	0	0	0.11	0	0
Chloropsis cyanopogon	NIF	0	0	0	0	0	0	0	0	0.05	0	0	0.11	0	0
Chloropsis sonnerati	NIF	0	0	0	0	0	0	0	0.05	0	0	0	0.06	0	0
Chloropsis cochinchinensis	NIF	0	0	0	0	0	0	0	0	0	0.18	0.02	0.33	0.11	0
Anthreptes simplex	NIF	0	0	0	0.02	0	0	0	0	0	0	0	0	0	0
Anthreptes malacensis	NIF	0.06	0.03	0	0.04	0.03	0	0	0	0	0	0.02	0	0.21	0
Nectarinia sperata	NIF	0	0	0	0.19	0	0.04	0	0	0	0	0	0	0	0.28
Arachnothera flavigaster	NIF	0	0	0	0	0	0	0	0	0	0	0	0.17	0	0
Dicaeum trigonostigma	NIF	0.19	0.07	0.11	0.13	0.3	0.54	0.22	0.05	0	0.29	0.09	0.06	0.26	0.56
Pernis ptilorhyncus	R	0	0	0.03	0	0	0.02	0	0	0	0.07	0	0	0	0
Spizaetus nanus (VU)	R	0	0	0	0	0	0	0	0.03	0	0	0	0	0	0
Microhierax fringillarius	SI	0	0.02	0	0	0.02	0.04	0	0	0.05	0	0	0	0	0
Surniculus lugubris	SI	0	0	0	0	0	0	0	0	0	0.04	0	0	0	0
Merops viridis	SI	0	0.05	0	0	0.02	0	0	0.26	0	0	0	0	0	0
Eurystomus orientalis	SI	0.06	0	0	0	0	0.15	0	0	0.2	0	0	0.11	0	0
Hemipus hirundinaceus	SI	0.03	0	0	0.09	0.16	0.04	0.22	0	0	0.21	0	0.33	0.32	0.06
Muscicapa dauurica	SI	0	0	0	0	0.03	0	0	0	0	0	0	0	0	0
Eumyias indigo	SI	0	0	0	0	0	0	0	0	0	0	0	0	0.05	0
Ficedula westermanni	SI	0	0	0	0	0	0	0	0	0	0	0.04	0	0	0
Cyornis turcosus	SI	0	0	0	0	0	0.02	0	0	0	0	0	0	0	0.06
Rhipidura javanica	SI	0.22	0	0.08	0.09	0.06	0.02	0	0	0	0	0	0	0	0
Hypothymis azurea	SI	0	0	0	0.04	0	0.07	0	0	0	0.21	0	0.17	0.32	0
Terpsiphone paradisi	SI	0	0	0	0	0	0.02	0	0	0	0	0	0	0	0
Artamus leucorynchus	SI	0.14	0	0.03	0	0	0	0	0	0	0	0	0	0	0
Eurylaimus ochromalus	SSGI	0	0	0	0	0	0.02	0	0	0	0.04	0	0	0	0
Dicrurus macrocercus	SSGI	0	0	0	0	0	0.04	0	0	0	0	0	0	0	0
Dicrurus aeneus	SSGI	0	0	0	0	0	0	0	0	0	0	0.04	0	0	0
Dicrurus remifer	SSGI	0	0	0	0.06	0	0.02	0.11	0.03	0	0.14	0	0.06	0	0
Dicrurus paradiseus	SSGI	0.03	0	0	0.07	0.02	0.06	0	0	0	0.11	0.04	0.11	0.05	0
Rhinomyias umbratilis	SSGI	0	0	0	0	0	0	0	0	0	0.07	0	0	0	0
Philentoma pyrhopterum	SSGI	0	0	0	0	0	0	0	0	0	0	0	0	0.11	0
Lanius tigrinus	SSGI	0	0	0	0	0	0	0	0.03	0	0	0	0	0	0
Macropygia ruficeps	TF	0.03	0	0	0	0	0	0	0	0	0	0	0	0	0
Chalcophaps indica	TF	0.03	0.02	0.03	0.04	0	0	0	0	0	0	0.02	0	0	0
Centropus bengalensis	TI	0	0	0	0	0.05	0	0	0.05	0.05	0	0.02	0	0	0
Pitta guajana	TI	0	0	0	0	0	0	0	0	0	0	0	0	0	0.22
Pellorneum capistratum	TI	0	0	0	0	0	0	0	0	0	0	0	0	0.16	0
Trichastoma rostratum	TI	0	0	0	0	0	0	0	0	0	0	0	0	0	0.17
Gallus gallus	TIF	0.06	0	0	0	0	0.02	0	0	0	0	0	0	0.26	0.06
Streptopelia chinensis	TIF	0.03	0.13	0.03	0	0	0.02	0	0	0	0	0	0	0	0
Geopelia striata	TIF	0	0	0	0	0	0	0	0.05	0	0	0	0	0	0
pigeon sp.	AF	0	0.02	0	0	0	0.02	0	0.03	0	0	0.02	0	0	0
babbler sp.	AFGI	0	0	0	0.04	0.08	0	0	0	0	0	0.02	0	0	0.11
malkoha sp.	AFGI	0	0	0	0	0	0	0	0	0	0	0	0	0.05	0
Orthotomus sp.	AFGI	0.11	0	0.03	0.07	0	0	0	0	0	0	0	0	0	0
Prinia sp.	AFGI	0	0	0.03	0	0	0.02	0	0	0.05	0	0	0	0	0
warbler sp.	AFGI	0	0	0	0	0.03	0	0	0	0	0	0	0	0	0
Pycnonotus sp.	AFGIF	0	0.02	0	0.02	0.21	0	0	0.16	0	0	0.11	0.06	0	0
woodpecker sp.	BGI	0	0	0	0.02	0	0.07	0	0.03	0.05	0.04	0	0	0.05	0
kingfisher sp.	MIP	0	0	0.03	0	0	0.02	0	0	0	0	0	0	0	0
spiderhunter sp.	NI	0	0	0.03	0	0	0	0	0	0.05	0	0	0	0	0
sunbird sp.	NI	0.06	0	0.14	0.11	0.03	0.02	0	0	0	0	0	0.06	0.05	0
flowerpecker sp.	NIF	0	0	0	0.04	0.03	0	0	0.03	0	0	0.02	0.06	0	0

leafbird sp.	NIF	0	0.02	0	0	0	0	0	0	0.15	0	0	0	0	0.06
raptor sp.	R	0	0	0	0	0	0.02	0	0	0	0.04	0	0	0	0
flycatcher sp.	SI	0	0	0.03	0	0	0.02	0	0	0	0	0.02	0	0	0
Rhipidura sp.	SI	0	0	0	0	0.02	0	0	0	0	0	0	0	0	0
drongo sp.	SSGI	0	0	0	0.04	0	0.02	0	0	0	0.07	0.04	0.22	0	0
Lanius sp.	SSGI	0	0	0	0	0	0	0	0	0	0	0.02	0	0	0

[1] Scientific name followed the nomenclature of Sibley & Monroe (1990).

[2] Abbreviations of feeding guilds shown here are: R, raptor; TI, terrestrial insectivore; AFGI, arboreal foliage-gleaning insectivore; BGI, bark-gleaning insectivore; SSGI, sallying substrate-gleaning insectivore; SI, sallying insectivore; AI, aerial insectivore; AFGIF, arboreal foliage-gleaning insectivore–frugivore; AF, arboreal frugivore; AFP, arboreal frugivore–predator; TF, terrestrial frugivore; TIF, terrestrial insectivore–frugivore; NI, nectarivore–insectivore; NF, nectarivore–frugivore; NIF, nectarivore–insectivore–frugivore; MIP, miscellaneous insectivore–piscivore. Classification of feeding guilds is based on Lambert (1992).

[3] Censuses were done for 10 minutes, and species inside a 25-m radius were recorded.

[4] Abbreviations for habitat types are: AP1, 1-year-old acacia plantation; AP4, 4-year-old acacia plantation; BLF, burnt logged forest; LLF, large logged forest; RLF, remnant logged forest.