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

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19 **Abstract**

20 Industrial timber plantations severely impact biodiversity in Southeast Asia. Forest fragments
21 survive within plantations, but their conservation value in highly deforested landscapes in
22 Southeast Asia is poorly understood. In this study, we compared bird assemblages in acacia
23 plantations and fragmented forests in South Sumatra to evaluate each habitat's potential
24 conservation value. To clarify the impact of habitat change, we also analyzed the response of
25 feeding guild composition. Five habitat types were studied: large logged forest (LLF), burnt
26 logged forest (BLF), remnant logged forest (RLF), 4-year-old acacia plantation (AP4), and
27 1-year-old acacia plantation (AP1). Estimated species richness (Chao 2) was highest in LLF
28 then AP4 and BLF, while AP1 and RLF had lower estimated species richness. Community
29 composition was roughly divided into two groups by non-metric multidimensional scaling
30 ordination: acacia plantation and logged forest. Sallying substrate-gleaning insectivores, such as
31 drongos, broadbills, and some flycatchers, were restricted to LLF, whereas acacia plantation
32 hosted many terrestrial frugivores, such as doves. Although fragmented forests in our study site
33 lacked several common tropical forest species, these fragments provide an important habitat for
34 some sallying and terrestrial insectivores. A network of small riparian remnant forests could be a
35 complementary habitat for some species, while the conservation value of burnt forest might be
36 low. In conclusion, the highly fragmented forests in plantations are suboptimal habitats for birds

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

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42 **Key Words**

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

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46

47 Introduction

48

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

60 Several studies have indicated that industrial timber plantations have severe impacts on
61 biodiversity (Barlow et al. 2007b), but only a few have examined acacia plantations (soil
62 macrofauna, Tsukamoto and Sabang 2005; beetles, Chung et al. 2000; mammals, Nasi et al.
63 2008 and McShea et al. 2009; birds, Styring et al. 2011). *Acacia mangium* is an important
64 industrial tree species and grows rapidly even on wasteland (Cossalter and Pye-Smith 2003).

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

76 One common feature of deforested and converted landscapes is forest fragmentation
77 (Laurance and Laurance 1999; Lindenmayer et al. 2002). Forest fragmentation is known to
78 decrease biodiversity in many tropical regions, and some authors submit that the conservation of
79 small forest fragments is less important than large fragments (Beier et al. 2002; Edwards et al.
80 2010). Hill et al. (2011) showed that the bird community is highly nested in small fragments
81 compared to that of insects, thus lowering the conservation value of small fragments for birds.
82 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

101 included in the analyses.

102

103 **Methods**

104

105 *Study Site*

106 The concession area of the PT. Musi Hutan Persada (PT. MHP) company is located in Muara

107 Enim District, South Sumatra Province, Indonesia (3°00'–4°00' S, 103°00'–104°30' E, Figure 1).

108 The topography is hilly, with an elevation of 60 to 200 m a.s.l. and sedimentary soil. In 2008,

109 when we performed our survey, *A. mangium* was planted on 190,000 ha of the 260,000 ha in the

110 concession area, which is divided into three parts: Wilayah I (region M), Wilayah II (region T),

111 and Wilayah III (not studied). Annual rainfall levels in 2008 were 2,008 mm in region M and

112 2,849 mm in region T, and mean annual temperature ranges were 25.5°C–28.0°C and

113 25.4°C–27.9°C, respectively (Shiotani et al. 2009). The dry season lasts from June until

114 September, and the wet season is from November until April. Approximately 95% of the

115 production area is planted with *Acacia mangium*, along with smaller plantations of *Eucalyptus*

116 *urophylla*, *Pinus merkusii*, *Paraserianthes falcataria*, *Gmelina arborea*, and other species. Trees

117 are harvested on a 6-year rotation and processed as pulp. In 2008, the second-rotation trees were

118 harvested, followed by incremental planting of the third-rotation trees. The plantation was

119 established in 1991 on former “alang-alang” (*Imperata cylindrica*) grasslands, scrublands, and
120 logged areas (Sireger et al. 1998). Based on historical vegetation maps, an estimated one third
121 of the concession area consisted of natural forest in 1985 (WWF-Indonesia 2010), while
122 approximately 80% was secondary forest or forest reserve in 1950 (Hannibal 1950). Thus, the
123 area experienced rapid forest loss between 1950 and 1985, before plantations were established.
124 This deforestation is consistent the estimate of Laumonier (1997), who reported an annual
125 deforestation rate of 1.6% between 1978 and 1985 in southwestern Sumatra, 30 km from our
126 study site. Typical forest transformation in this region occurred first through logging, followed
127 by the allocation of land to transmigration programs or industrial plantations (Laumonier 1997).
128 As a result, South Sumatra province saw abandoned *Imperata* grassland spread over 708,000 ha
129 in the 1990s, more than 6% of the province’s area (Garrity et al. 1997).

130 Following a new Indonesian law requiring that approximately 10% of the concession area
131 be set aside for conservation (Keputusan Menteri Kehutanan No.70/Kpts-II/95), 25,775 ha, or
132 9.9% of the total, is now reserved. A network of remnant riparian forests remains mainly along
133 streams and rivers, following a separate regulation aimed at protecting riparian buffers
134 (Peraturan Pemerintah Republik Indonesia No.38/2011 Tentang Sungai), although the
135 conservation area is larger than these riparian forest remnants. However, illegal logging
136 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*

155 3–5 m in height was the only major plant species. In region M, the three LLF fragments
156 consisted of trees 20 m tall with multistory vegetation, while the one BLF fragment was
157 disturbed and invaded by many acacia trees and ferns. The AP4, AP1, and remnant RLF points
158 in region M were similar to those in region T.

159 We generated a vegetation map (Figure 1) by combining satellite images from ALOS
160 AVNIR-2 (Advanced Land Observing Satellite “DAICHI”, Advanced Visible and Near Infrared
161 Radiometer type 2) (April 11–May 10, 2008 and October 18, 2010) with the land cover map of
162 Miettinen et al. (2012). The normalized difference vegetation index (NDVI) was calculated
163 from ALOS images, followed by supervised classification using maximum likelihood. Oil palm
164 plantations were masked to exclude them from the classification, because of the potential to
165 confuse them with acacia stands. The coarser land cover map of Miettinen et al. (2010) was
166 included to compensate for masked portions of the ALOS images or those covered by clouds.
167 Of the original categories, “lowland forest” and “lower montane forest” were recategorized as
168 “Woodland/Forest,” “plantation/regrowth” as “Acacia/Regrowth,” and “lowland mosaic” and
169 “lowland open” as “Shrub/Open Land.” Satellite image analysis and mapping were performed
170 with ENVI ver. 4.8 (Exelis VIS, Boulder, CO, USA) and ArcGIS 10 (Esri, Redlands, CA, USA)
171 software. Because of the similar NDVI values among acacia plantations, oil palm plantations,
172 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

191 bird specialist (M. Iqbal, W. Satrio, M. Dwi, and N. Wilson). Bird songs and calls were recorded
192 on an IC-recorder to identify uncertain species later in the lab. We used “Birds of Tropical Asia
193 3.0” software (Scharringa 2005) to identify unknown bird songs. Individuals that could be
194 identified only to the group level were recorded as “*group name.sp.*”

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

199

200 *Statistical analyses*

201 We calculated species rarefaction and extrapolation using EstimateS ver. 9.1.0 software
202 (Colwell 2013). As sampling effort differed among habitat types and the species accumulation
203 curve was generally not saturated, comparing species richness using the raw data was difficult.
204 To overcome these problems and to compare species richness more reliably, we extrapolated
205 and calculated estimators. Extrapolation is used to compare smaller with larger samples using
206 statistical sampling models rather than functional curve fitting (Colwell et al. 2012). We
207 calculated up to 200 samples using 95% confidence intervals. Richness estimators (Chao 2,
208 Jackknife 1, and Jackknife 2) were also calculated. In all, 132, 171, 67, 93, and 37 census

209 samples were analyzed for AP1, AP4, BLFs, LLFs, and remnant RLFs, respectively. Individuals
210 that flew over the census point were included in the analyses, but those identified only to the
211 group level were excluded. The Chao 2 richness estimator approaches asymptotic species
212 richness if the sample size is sufficiently large; that is, if the number of unique observations
213 relative to the total number of observations is less than 50% (EstimateS 9.1.0 User's Guide).
214 Our dataset met this condition; therefore, each sample size was large enough to estimate species
215 richness.

216 Sixteen feeding guilds were classified based on Lambert (1992): R, raptor; TI, terrestrial
217 insectivore; AFGI, arboreal foliage-gleaning insectivore; BGI, bark-gleaning insectivore; SSGI,
218 sallying substrate-gleaning insectivore; SI, sallying insectivore; AI, aerial insectivore; AFGIF,
219 arboreal foliage-gleaning insectivore–frugivore; AF, arboreal frugivore; AFP, arboreal
220 frugivore–predator; TF, terrestrial frugivore; TIF, terrestrial insectivore–frugivore; NI,
221 nectarivore–insectivore; NF, nectarivore–frugivore; NIF, nectarivore–insectivore–frugivore; and
222 MIP, miscellaneous insectivore–piscivore. We also calculated individual-based bird occurrence
223 per census for each feeding guild at each habitat type. Individuals that flew over the census
224 point were excluded from the analyses. Individuals identified only to the group level were
225 included in the analysis by classifying them into the feeding guild in which most species of the
226 same group were categorized. For example, unidentified drongos was classified as SSGI,

227 because most species (e.g., *Dicrurus paradiseus*, *D. remifer*, and *D. aeneus*) belonged to this
228 category. To analyze the deviation of occurrence in each habitat type and feeding guild category,
229 we calculated the squared difference between observed and expected data (deviation, d_{ij} ; the
230 same procedure as in the χ^2 test) as follows:

$$231 \quad d_{ij} = \frac{(O_{ij} - E_{ij})^2}{E_{ij}} \quad (\text{Equation 1})$$

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

234 To analyze differences in community composition among habitat types, we performed an
235 ordination of study sites against density data for each survey year and habitat type, using
236 non-metric multidimensional scaling (NMDS) with the package “vegan” (Oksanen et al. 2010)
237 in R version 3.0.2 (R Development Core Team 2013). NMDS was performed with the
238 Bray–Curtis dissimilarity index and a maximum of 100 iterations to obtain better ordination
239 scores. Correlations between the NMDS axes and environmental variables, such as habitat type
240 (AP1/AP4, BLF/LLF/RLF), vegetation cover (%) at different tree/shrub heights (0–1, 1–10,
241 10–20, and 20–30 m), and distance to the nearest conservation area (km), were also calculated.
242 We also calculated the species score against each axis, although only abundant species were
243 labeled when several species appeared in close proximity. Individuals that flew over the census
244 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

263 (74.4), AP1s (61.1), and remnant RLFs (50.0) (Table 1). However, the differences among Chao
264 2 values were not statistically significant, as their respective 95% confidence intervals showed
265 high degrees of overlap. The other estimators, Jackknife 1 and Jackknife 2, yielded similar
266 results, although AP1s and RLFs were reversed in order. Overall bird density (per census
267 abundance) in natural forests (6.2 in RLF; 4.3 in BLF; 3.4 in LLF) was higher than in acacia
268 plantations (3.5 in AP4; 1.9 in AP1) (Table 1). These differences could be even more marked
269 considering the likelihood of underestimating the number of birds in natural forest because of
270 the limited visibility compared with more open acacia plantations. We observed one “vulnerable”
271 species (*Spizaetus nanus*) in BLF in 2007 (BirdLife International 2012; Appendix 2).

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

278 AP4s were home to arboreal foliage gleaning insectivore-frugivores (AFGIF),
279 including medium-sized bulbuls, especially *Pycnonotus goiavier*, and
280 smallnectarivore-insectivore-frugivores (NIF), such as *Dicaeum trigonostigma* and *Nectarinia*

281 *sperata*, but we observed fewer arboreal foliage gleaning insectivores (AFGI) (Table 2). In
282 contrast, AP1s were characterized by more terrestrial insectivore-frugivores (TIF), including
283 *Streptopelia chinensis* and *Geopelia striata*, and terrestrial frugivores (TF), such as
284 *Chalcophaps indica*, but we observed fewer NIFs. Fewer sallying insectivores (SI) and sallying
285 substrate gleaning insectivores (SSGI) were seen in acacia plantations, except for some forest
286 edge species, including *Hemipus hirundinaceus*, *Eurystomus orientalis*, and *Rhipidula javanica*.
287 These results showed that community structure develops according to the age of the acacia
288 trees.

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

299 The NMDS ordination of survey points and subsequent ANOSIM indicated significant
300 differences in community assemblages among habitat types and years but not regions (Figure
301 4a; Table 3). The similarity in community structure between regions T and M indicated that they
302 can be treated as replicates. Among habitat types, AP1s (0.190) and BLF fragments (0.134)
303 showed higher NMDS 1 values than LLF (−0.214) or remnant RLF fragments (−0.242)
304 (Table 4). We found lower NMDS 2 values for LLF fragments (−0.200) and BLF fragments
305 (−0.170) than for AP1s (0.113) or AP4s (0.074). NMDS 1 correlated positively with shrub cover
306 (vg.0110) and negatively with cover of trees 10–20 m tall (vg.1020). NMDS 2 correlated
307 positively with distance from the nearest conservation area and negatively with understory
308 vegetation cover (vg.0001) and cover of trees 20–30 m tall (vg.2030).

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

317

318 **Discussion**

319

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

325 Our stand-scale survey showed that LLF and RLF fragments have great value in
326 conserving bird diversity. The community compositions of AP1s and AP4s were different from
327 those in logged forests, including LLF, BLF, and RLF fragments (Table 4). The most
328 species-rich LLF fragment was characterized by many forest species that occurred only in such
329 habitats (Figure 4b). Nevertheless, compared to the ordination results of Styring et al. (2011),
330 differences between natural logged forests and acacia plantations were less clear in our study
331 area with respect to species number, feeding guild pattern, and community structure. That is,
332 AP4s showed a community composition that was closer to those of the logged forests in NMDS
333 1 (Table 4; Figure 4a); AP4s included some species that occurred in logged forests, such as
334 *Pycnonotus goiavier*, *Pycnonotus plumosus*, and *Pycnonotus atriceps* (Figure 4b), which favor

335 open forests and forest edges.

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

344 Although ordination analysis did not detect significant differences in community
345 structure between RLFs and LLFs, there were differences in both total species density and
346 feeding guilds. The feeding guild pattern in LLF fragments showed fewer representatives of
347 arboreal foliage gleaning insectivores (AFGI) and more of arboreal foliage gleaning
348 insectivore-frugivores (AFGIF), whereas RLFs showed the opposite pattern, with fewer AFGIF
349 species, such as *Pycnonotus* spp., and more AFGIs, especially ioras, *Macronous gularis*,
350 *Orthotomus ruficeps*, and *Orthotomus atrogularis*. RLF was also characterized by more
351 terrestrial insectivore (TI) species, especially *Pitta guajana*, *Pellorneum capistratum*, and
352 *Trichastoma rostratum*. These observations indicated that RLFs are more suitable for small

353 insectivores that feed mostly along the forest edge and on the forest floor. LLFs were
354 characterized by more SSGI species, including drongos, *Eurylaimus ochromalus*, and
355 *Rhinomyias umbratilis*. Hornbills were also seen mostly in LLF fragments, which would be
356 suitable for medium-sized fly-catching insectivores and bulbuls.

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

368 Compared to other studies (Danielsen and Heegaard 1995; Thiollay 1995) in the 1990s in
369 lowland forest on Sumatra, many forest species, such as trogons, broadbills, barbets, and
370 babblers, were absent from our study site. In contrast, we found species that favor open habitats

371 or the forest edge, such as warblers, doves, and minivets, which were not recorded in those
372 previous studies. These species were seen in both logged forest and acacia plantations,
373 indicating that the logged forest fragments at our study site do not harbor many species
374 restricted to dense tropical rain forest, the original vegetation on this island. As a result of
375 extensive forest fragmentation across South Sumatra province in the 1970s due to intensive
376 cultivation and logging, this area is now only inhabited by species adapted to fragmentation.

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

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

400 The results of the present study suggested that fragmented natural forests harbor richer
401 bird diversity than plantations, a result that is consistent with previous reports (Najera and
402 Simonetti 2009; Edwards et al. 2010). Although fragmented natural forests do not help to
403 conserve primary forest species that would have been present in the past or in other regions,
404 they can harbor other species that are resistant to habitat modification. Fragmented forests play
405 an important role in biodiversity conservation in this region, where large primary forest blocks
406 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.

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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

569 plantation; BLF, burnt logged forest; LLF, large logged forest; RLF, remnant logged forest.
570 Abbreviations for feeding guilds shown here are: R, raptor; TI, terrestrial insectivore; AFGI,
571 arboreal foliage-gleaning insectivore; BGI, bark-gleaning insectivore; SSGI, sallying
572 substrate-gleaning insectivore; SI, sallying insectivore; AI, aerial insectivore; AFGIF, arboreal
573 foliage-gleaning insectivore–frugivore; AF, arboreal frugivore; AFP, arboreal
574 frugivore–predator; TF, terrestrial frugivore; TIF, terrestrial insectivore–frugivore; NI,
575 nectarivore–insectivore; NF, nectarivore–frugivore; NIF, nectarivore–insectivore–frugivore;
576 MIP, miscellaneous insectivore–piscivore. Classification of feeding guilds is based on Lambert
577 (1992).

578

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

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

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

600

601

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

	Habitat type				
	AP1	AP4	BLF	LLF	RLF
<i>Species richness and diversity</i>					
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
<i>Estimators</i>					
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
<i>General information</i>					
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

603 ¹ Abbreviations for habitat types are: AP1, 1-year-old acacia plantation; AP4, 4-year-old acacia

604 plantation; BLF, burnt logged forest; LLF, large logged forest; RLF, remnant logged forest.

605 ² Uniques are species that occurred in only one sample.

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

607

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

Feeding guild ²	Habitat type ¹				
	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

611

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

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

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

- 617 substrate-gleaning insectivore; SI, sallying insectivore; AI, aerial insectivore; AFGIF, arboreal
- 618 foliage-gleaning insectivore–frugivore; AF, arboreal frugivore; AFP, arboreal
- 619 frugivore–predator; TF, terrestrial frugivore; TIF, terrestrial insectivore–frugivore; NI,
- 620 nectarivore–insectivore; NF, nectarivore–frugivore; NIF, nectarivore–insectivore–frugivore;
- 621 MIP, miscellaneous insectivore–piscivore.
- 622
- 623

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

	Statistic <i>R</i>	<i>P</i> value
habitat (5 groups)	0.3260	0.001
year (2 groups)	0.1852	0.01
region (2 groups)	0.0795	0.067

625

626

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

Habitat type ²	NMDS1 ³			NMDS2		
	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

628 ¹ No data available.

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

631 ³ NMDS refer to non-metric multidimensional scaling.

632

Figures

Figure 1

639 **Figure 2**

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644 **Figure 3**

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648 **Figure 4**

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652 **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 plantation (AP1)	acacia	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 plantation (AP4)	acacia	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 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

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

Scientific name	Feeding guild	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
<i>Treron oxyura</i>	AF	0	0	0.03	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Treron curvirostra</i>	AF	0	0	0	0	0	0	0	0	0	0.05	0	0	0	0	0	0	0
<i>Treron vernans</i>	AF	0	0.07	0	0	0.08	0.09	0	0.05	0	0	0	0	0	0	0	0	0.11
<i>Irena puella</i>	AF	0	0	0	0	0	0	0	0.03	0	0	0	0	0.11	0	0	0	0
<i>Gracula religiosa</i>	AF	0	0	0.03	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Cacomantis sonneratii</i>	AFGI	0	0	0	0	0	0	0	0	0	0	0	0.02	0	0	0	0	0
<i>Cacomantis merulinus</i>	AFGI	0	0	0	0	0.02	0	0	0.03	0	0	0	0	0	0	0.05	0	0
<i>Phaenicophaeus diardi</i>	AFGI	0	0	0.03	0	0	0	0	0	0	0	0	0	0	0	0.05	0	0
<i>Phaenicophaeus chlorophaeus</i>	AFGI	0	0	0	0	0	0	0	0	0	0	0	0	0	0.11	0	0	0
<i>Phaenicophaeus curvirostris</i>	AFGI	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.17
<i>Sasia abnormis</i>	AFGI	0	0	0	0	0	0.02	0	0	0	0.04	0	0	0	0	0	0	0
<i>Dendrocopos canicapillus</i>	AFGI	0	0	0	0	0	0	0	0	0.05	0	0	0	0	0	0	0	0
<i>Dendrocopos moluccensis</i>	AFGI	0	0	0	0.02	0.02	0.02	0	0	0	0	0	0	0	0	0	0	0
<i>Tephrodornis gularis</i>	AFGI	0	0	0	0	0	0	0	0	0	0	0	0.02	0.06	0	0	0	0
<i>Coracina fimbriata</i>	AFGI	0	0	0	0	0	0	0	0.03	0	0.07	0	0	0	0	0.05	0	0
<i>Pericrocotus igneus</i>	AFGI	0	0	0	0	0	0	0	0	0.11	0.2	0	0.04	0	0.11	0	0	0
<i>Pericrocotus solaris</i>	AFGI	0	0	0	0	0	0	0.11	0	0	0	0	0	0	0	0	0	0
<i>Aegithina viridissima</i>	AFGI	0	0	0	0.02	0	0	0	0	0	0	0	0	0	0	0.16	0.11	0
<i>Aegithina tiphia</i>	AFGI	0	0	0.03	0.06	0.06	0.04	0.11	0	0	0	0	0	0.06	0.47	0	0	0
<i>Malacocincla sepium</i>	AFGI	0.06	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Stachyris erythroptera</i>	AFGI	0	0	0	0	0	0.06	0	0	0	0	0	0	0	0	0	0	0
<i>Macronous gularis</i>	AFGI	0.17	0	0.08	0.15	0	0.15	0	0	0.2	0	0	0.06	0	0	0	1.33	0
<i>Copsychus saularis</i>	AFGI	0.06	0.03	0	0.06	0.05	0	0	0	0	0	0.02	0	0	0	0	0	0
<i>Phylloscopus inornatus</i>	AFGI	0	0	0	0	0	0	0	0	0	0.04	0	0	0	0	0	0	0
<i>Orthotomus atrogularis</i>	AFGI	0	0	0	0	0	0	0.22	0	0	0.14	0	0	0.58	0.22	0	0	0
<i>Orthotomus ruficeps</i>	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	0	0	0
<i>Orthotomus sericeus</i>	AFGI	0.08	0	0.08	0.09	0	0.17	0	0	0	0.04	0	0.22	0.11	0.44	0	0	0
<i>Prinia flaviventris</i>	AFGI	0.25	0.02	0.14	0	0.08	0.07	0	0.03	0.35	0.04	0	0	0	0	0	0	0
<i>Prinia familiaris</i>	AFGI	0	0.05	0	0	0.02	0	0	0.11	0.1	0	0	0	0	0	0	0	0
<i>Pachycephala grisola</i>	AFGI	0	0	0	0.07	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Calorhamphus fuliginosus</i>	AFGIF	0	0	0	0	0	0	0	0	0.1	0	0	0	0	0	0	0	0
<i>Pycnonotus atriceps</i>	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	0	0	0
<i>Pycnonotus melanicterus</i>	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	0	0	0
<i>Pycnonotus cyaniventris</i>	AFGIF	0	0	0	0	0	0	0	0	0	0	0	0.17	0	0	0	0	0
<i>Pycnonotus aurigaster</i>	AFGIF	0.33	0.03	0.22	0.13	0.21	0.24	0.11	0	0.65	0	0	0	0	0	0	0	0
<i>Pycnonotus goiavier</i>	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	0	0	0
<i>Pycnonotus plumosus</i>	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	0	0	0
<i>Pycnonotus simplex</i>	AFGIF	0	0	0	0.02	0	0.02	0	0	0	0.04	0	0.5	0	0.06	0	0	0
<i>Pycnonotus brunneus</i>	AFGIF	0	0	0	0.02	0.11	0.28	0	0.29	0.25	0.29	0.28	0.17	0	0.22	0	0	0
<i>Pycnonotus erythrophthalmos</i>	AFGIF	0	0	0	0	0.02	0	0	0	0	0.14	0.04	0.17	0	0.11	0	0	0
<i>Alophoixus bres</i>	AFGIF	0	0	0	0	0	0	0	0	0	0.07	0	0.11	0	0	0	0	0
<i>Tricholestes criniger</i>	AFGIF	0	0	0	0.02	0	0	0	0	0	0.04	0	0.17	0	0.11	0	0	0
<i>Ixos malaccensis</i>	AFGIF	0	0	0	0	0	0	0	0	0	0.11	0	0	0	0	0	0	0
<i>Corvus enca</i>	AFGIF	0	0	0	0	0.02	0	0	0	0	0	0.04	0	0	0	0	0	0
<i>Anthracoceros malayanus</i>	AFP	0	0	0	0.04	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Buceros rhinoceros</i>	AFP	0	0	0	0	0	0	0	0	0	0	0	0.06	0	0	0	0	0
<i>Hemiprocne longipennis</i>	AI	0	0	0	0	0.05	0.06	0	0	0	0	0	0	0	0	0	0	0
<i>Hirundo rustica</i>	AI	0	0	0.14	0	0	0	0	1.24	0	0	0	0	0	0	0	0	0
<i>Celeus brachyurus</i>	BGI	0	0	0	0	0	0	0	0	0	0.07	0	0	0	0	0	0	0
<i>Picus mineaceus</i>	BGI	0	0	0	0	0	0.02	0	0	0	0	0	0	0	0	0	0	0
<i>Blythipicus rubiginosus</i>	BGI	0	0	0	0.02	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Reinwardtipicus validus</i>	BGI	0	0	0	0	0	0	0	0	0	0	0	0.11	0	0	0	0	0
<i>Sitta frontalis</i>	BGI	0.08	0	0	0.04	0	0	0	0	0	0	0	0	0	0	0	0	0

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

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

656 ² Abbreviations of feeding guilds shown here are: R, raptor; TI, terrestrial insectivore; AFGI, arboreal foliage-gleaning

657 insectivore; BGI, bark-gleaning insectivore; SSGI, sallying substrate-gleaning insectivore; SI, sallying insectivore; AI,

658 aerial insectivore; AFGIF, arboreal foliage-gleaning insectivore–frugivore; AF, arboreal frugivore; AFP, arboreal

659 frugivore–predator; TF, terrestrial frugivore; TIF, terrestrial insectivore–frugivore; NI, nectarivore–insectivore; NF,

660 nectarivore–frugivore; NIF, nectarivore–insectivore–frugivore; MIP, miscellaneous insectivore–piscivore.

661 Classification of feeding guilds is based on Lambert (1992).

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

663 ⁴ Abbreviations for habitat types are: AP1, 1-year-old acacia plantation; AP4, 4-year-old acacia plantation; BLF, burnt

664 logged forest; LLF, large logged forest; RLF, remnant logged forest.