

1 **Impacts of land-use change on endemic avifauna on Makira, Solomon Islands: endemics**
2 **avoid monoculture**

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14 **Running head:** Impacts of land-use change on tropical avifauna

15 **Abstract**

16 Unprecedented rates of deforestation on tropical islands are threatening high numbers of endemic
17 species. More empirical evidence is needed to better understand the implications of land-use
18 change on biodiversity, and to guide conservation actions. We assessed the impacts of land-use
19 change on the lowland avifauna of the tropical island of Makira in the Solomon Islands. We
20 examined species richness and community assemblages, with a particular focus on endemism and
21 functional traits, to provide further insight into the 'conservation value' of the dominant land-use
22 types present on Makira (i.e. intact forest, secondary forest, food gardens, mixed cocoa plantations,
23 and monoculture cocoa plantations). We found species richness was similar across habitats, but
24 endemic species richness decreased as intensity of land use increased. There were significant
25 differences in the occurrence of functional groups between habitats. Fifteen out of the 42 species
26 observed showed significant variation in abundance across habitats. Of those species that varied, 12
27 were endemic to Makira or to Melanesia, with seven of these endemics being absent from the
28 monoculture cocoa. As tropical islands have less functional redundancy than continental
29 landmasses, protecting the remaining forest and improving habitat connectivity will be even more
30 critical for conserving their endemic species and maintaining ecosystem functioning.

31

32 **Introduction**

33 Tropical forests cover just 7% of the Earth's land surface, but support more than 60% of known
34 species (Bradshaw *et al.* 2009; Laurance 1999; Wright 2005). These forests provide essential
35 ecosystem services, such as carbon storage, clean water provision and contributions to climate
36 stability, playing a major role in human well-being (Tobias *et al.* 2013). However, tropical
37 deforestation is continuing at unprecedented rates with 40 million hectares of primary forest lost
38 since 2000 (FAO 2010). This loss is largely driven by intensification of land use, which has led to
39 fragmentation and degradation of habitat (Swift and Hannon 2010). Habitat loss is particularly rapid
40 in developing regions where human populations are expanding and where the majority of people
41 directly depend on natural resources for subsistence (Msuha *et al.* 2012). Population growth rates in
42 these regions are substantially higher than global rates (Cincotta *et al.* 2000). With consumption
43 levels expected to exceed population growth (Sodhi *et al.* 2013), agricultural expansion will remain a
44 major driver of habitat loss and corresponding declines in biodiversity (Phalan *et al.* 2013).

45 Tropical islands are experiencing higher rates of deforestation than continental areas
46 (Achard *et al.* 2002), and thus are an increasingly important focus for conservation efforts. They also
47 support high numbers of endemic species with traits that make them particularly prone to
48 extinction, such as habitat and diet specialisation, and a restricted range (Aratrakorn *et al.* 2006;
49 Purvis *et al.* 2000; Senior *et al.* 2013). Understanding how island endemic species respond to land-
50 use change, and the degree to which tropical forest organisms can persist in human-dominated
51 landscapes can improve decision making for environmental management (Flynn *et al.* 2009; Gardner
52 *et al.* 2010). However, few studies have addressed impacts of land use change on tropical island
53 diversity (but see de Lima *et al.* 2012; Woinarski 2010) and the conservation implications of modified
54 island landscapes are poorly understood (Goldman *et al.* 2008; Woinarski 2010). High levels of
55 endemism across island biotas lend added urgency to understanding how land use changes impact
56 community composition, patterns of biodiversity, and extinction threats (de Lima *et al.* 2012;
57 Waltert *et al.* 2011; Woinarski 2010).

58 As a taxonomic group, birds are a logical entry point to understand the impact of land use
59 changes on biodiversity. Not only are birds comparatively well studied, diverse, and cost-effective to
60 sample (Barlow *et al.* 2007; Kessler *et al.* 2011), they also have vital roles in key ecosystem
61 processes, such as pollination, seed dispersal, nutrient cycling, pest control and scavenging (Wenny
62 *et al.* 2011; Whelan *et al.* 2008). Thus, bird population declines may lead to changes that cascade
63 through ecosystems and subsequently cause deterioration of ecosystem services provided to
64 humans (Wenny *et al.* 2011). Anthropogenic land use changes alter habitat structure and resource
65 provision, potentially favouring some species and disadvantaging others. Species traits both dictate
66 how species respond to environmental change and also influence their provision of ecosystem
67 services (Luck *et al.* 2012; Violle *et al.* 2007). It is increasingly recognised that it is functional diversity
68 rather than species diversity *per se* that is of greater importance to maintaining a range of
69 ecosystem services (Flynn *et al.* 2009). Species traits are key to understanding how particular species
70 affect ecosystem function (Hooper *et al.* 2005). Consequently, assessing how species with different
71 functional traits respond to land-use changes acts as a proxy for concurrent impacts on ecosystem
72 function (Hooper *et al.* 2005; Senior *et al.* 2013). Similarly, identifying species traits that correspond
73 to favourable or unfavourable responses to altered habitats can aid conservation efforts by
74 predicting species groups at greatest risk from land-use change (McGill *et al.* 2006).

75 In this study, we assess the impacts of land-use change on avifauna of the tropical island of
76 Makira, Solomon Islands. The Solomon Islands, located in the South West Pacific, present an ideal
77 study site because they are of global importance to biodiversity, are part of the East Melanesian
78 Islands biodiversity hotspot (Myers *et al.* 2000) and are a critical Endemic Bird Area, with more
79 restricted-range bird species than any other Endemic Bird Area (BirdLife International 2015). There
80 are a total of 293 bird species across the Solomon Islands, with 85 species endemic to the island
81 group (Dutson 2011). The geologically highly complex archipelagos of Melanesia have developed in
82 the transition between the Australo-Papuan plate and the Pacific plate (see Hall 2002), with
83 molecular studies suggesting that the diverse array of endemic island genera and species is a result

84 of extensive radiations of species within archipelagos and even back to the continents (Filardi and
85 Moyle 2005), and thus they appear to represent an incredible ‘factory’ in terms of global biological
86 diversification (Danielsen *et al.* 2010). Makira is separated from the rest of the Solomon Islands
87 archipelago by a deep water trench; it supports 18 extant endemic bird taxa, which is more
88 restricted-range bird species and subspecies than any other area of comparable size in the world
89 (Danielsen *et al.* 2010) and is an important area of endemism in its own right.

90 The Solomon Islands also contain one of the last remaining tracts of undisturbed coastal
91 tropical rainforest (Bayliss-Smith *et al.* 2003), but Makira, like the rest Solomon archipelago, is
92 undergoing rapid land use change. The tropical rainforests of the Solomon Islands have long been
93 influenced by natural disturbances, such as cyclones, and human impacts, including swidden
94 agricultural practices, which have created a mosaic of secondary forest successions in many lowland
95 forest areas (see Bayliss-Smith *et al.* 2003). However, it is the current rate and scale of change across
96 the region that is a primary concern, driven by an economy strongly biased towards extractive
97 industries, primarily logging (CBSI 2011), coupled with one of the highest human population growth
98 rates in the world (2.6% per annum; UNICEF 2011). Despite rapid rates of land conversion, few
99 studies have directly looked at the impacts of these changes on patterns of diversity. We examined
100 bird species richness and community assemblage, with a particular focus on endemism and
101 functional traits, to provide further insight into the ‘conservation value’ (Waltert *et al.* 2011) of the
102 five dominant habitats present within our study area on Makira. Specifically we aimed to quantify
103 the impacts of land-use change on the Makira avifauna by assessing variation, along a gradient of
104 land use intensity, in: 1) species richness, 2) community assemblage, including endemism and
105 functional traits (microhabitat and feeding guild), and 3) species-specific changes in abundance
106 (community structure).

107

108 **Methods**

109 *Study area*

110 Our study was focussed in the Kahua region of Makira province in the Solomon Islands (Figure 1).
111 Makira is the fourth largest island of the archipelago with an area of 3191 km², and consists of a
112 narrow coastal plain leading to steep forested central ridges, with elevations of up to 1200 m (Allen
113 *et al.* 2006). It has a wet tropical climate with little annual variation (Fasi *et al.* 2013). Makira
114 contains globally significant biodiversity with 12 species endemic to Makira and its small off-lying
115 islands Ugi and Santa Ana (Owaraha), and an additional six subspecies that are highly divergent and
116 would qualify as species if a phylogenetic species concept was employed (see Danielsen *et al.* 2010;
117 Dutson 2011). The majority of endemic species on Makira are thought to have a wide altitudinal
118 range. Exceptions are the Shade Warbler (*Cettia parens*), San Cristobal Leaf-warbler (*Phylloscopus*
119 *makirensis*), Makira Thrush (*Zoothera margaretae*), and Dusky Fantail (*Rhipidura tenebrosa*), which
120 are largely confined to highland areas (see Danielsen *et al.* 2010 for further details). However, there
121 is very little recorded knowledge on the natural history of the Makiran avifauna, their distribution on
122 altitudinal gradients nor their dependence on different habitats (Danielsen *et al.*, 2010). There have
123 been no prior surveys of the avifauna in Kahua, yet based on available habitat and topography, we
124 would expect all endemic species to be present within this region.

125 The Kahua region is one of the only areas on the island not to have any commercial logging
126 licenses (past or present) (Pauku 2009), and because logging companies are interested mainly in flat,
127 lowland areas, the steep terrain of the Kahua region could be considered a relatively safe sanctuary
128 for biodiversity. However, decreases in primary productivity have been detected in Kahua,
129 suggesting environmental change at a landscape scale (Garonna *et al.* 2009). One of the most
130 noticeable changes within the Kahua region is the proliferation of cocoa plantations. Cocoa was first
131 introduced to the Solomon Islands in 1950s, with extensive plantations undertaken by smallholders
132 in the late 1970s; presently, national policies and external aid continue to promote smallholder
133 production of cocoa throughout the Solomon Islands (Hivu 2013). Currently the larger cocoa
134 plantations in Kahua are grown on floodplain areas, but cocoa is not grown exclusively in these
135 areas. Local people report that these floodplain areas used to be used for subsistence gardens and

136 that since the introduction of cocoa, gardens have been moved further away to steeper-sided
137 slopes.

138

139 *Data collection*

140 Data were collected between January and July 2012. Two 500 m long transects were established in
141 each of the five land-use types in Kahua (n = 10; Figure 1). Topographically, all transects were
142 situated between 20 m and 500 m above sea level (e.g. within lowland forest) to avoid habitat
143 changes associated with higher altitudes. The sampled land-use types increased in land use intensity
144 and were classified as:

145 1) Intact forest: closed canopy (30-45 m high), comprising large hardwood trees, including those of
146 higher quality timber (e.g. Oceanic Lychee (*Pometia pinnata*), New Guinea Teak (*Vitex cofassus*),
147 Rosewood (*Pterocarpus indicus*)), with dense understory vegetation including thickets of smaller
148 trees, rattan palms (*Calamus* spp.), *Stenochlaena* ferns and *Selaginella* mosses. Anthropogenic
149 disturbance is a ubiquitous feature of the forests of the Solomon Islands (Bayliss-Smith *et al.* 2003),
150 as such no forest in this region can be considered “primary” in its truest sense. We therefore use
151 intact forest to refer to the lowland, evergreen tropical rainforest (0–500 m asl) with, historical, but
152 presently limited human disturbance.

153 2) Secondary forest: no continuous canopy, although crowns can be in close proximity to one
154 another, mainly composed of small fast growing, pioneer species, (including *Macaranga* spp., *Ficus*
155 spp., and *Hibiscus tiliaceus*) interspersed with larger trees, including Ngali nut (*Canarium indicum*),
156 breadfruit (*Artocarpus altilis*), coconut (*Cocos nucifera*) and sago palm (*Metroxylon salomonense*).
157 This habitat has never been commercially logged, but has historically and continues to be used
158 intensively by local communities for the collection of firewood, timber and wild plants.

159 3) Garden: an open canopy above food crops such as yam (*Dioscorea* spp.), taro (*Colocasia*
160 *esculenta*), sweet potato (*Ipomoea batatas*) and slippery cabbage (*Abelmoschus manihot*), as well as
161 various protected or deliberately planted herbaceous and tree species, such as coconut palms,

162 banana cultivars (*Musa* cultivars), sago palm, betel-nut palm (*Areca catechu*), nut trees (e.g.
163 *Canarium* spp., *Barringtonia edulis*, *Inocarpus fagifer*), and fruit trees (*Ficus* spp., *Atrocarpus altilis* ,
164 *Mangifera indica*, *Carica papaya*). Garden is the Melanesian term for land used for small-scale
165 agriculture (see Mertz *et al.* 2012); such areas are often located beyond the immediate vicinity of
166 dwellings and are not strictly 'homegardens' as defined elsewhere in the world.

167 4) Mixed-cocoa: smallholder cocoa plantations (*Theobroma cacao*), which typically grow up to 4-8 m.
168 Cocoa trees are planted close together resulting in a low closed canopy. There is no understory
169 because it is regularly cleared. Small patches of cocoa trees are regularly interspersed with coconut
170 palms and large fruit and nut trees, such as breadfruit and *Canarium* spp, with irregular patches of
171 low-canopy regenerative tree and herbaceous species, including gingers (Zingiberaceae) and betel-
172 nut.

173 5) Monoculture cocoa: extensive smallholder cocoa plantations, grown in flood plain areas with a 4-
174 8 m high closed canopy and no understory due to regular clearance. Monoculture cocoa is irregularly
175 interspersed with coconut trees and occasionally large lone trees, (1-2 per transect) such as
176 breadfruit or Malay apple (*Syzygium malaccense*), but with almost no shade cover.

177

178 Transects were traversed on foot. Every bird that was detected by sight or sound was recorded. The
179 perpendicular distance from the transect line was estimated at the time of detection (Bibby *et al.*
180 2000), by walking up to the point on the transect that was deemed perpendicular to the bird and the
181 distance estimated from this point. Calling birds were recorded as if visual sightings although extra
182 care was taken in estimating the distance from the observer. To maximize detection, surveys were
183 conducted between 06:00 – 08:00, usually within 30 minutes of sunrise; in appropriate weather
184 conditions (e.g. not in heavy rain and/or high wind). Transects were walked at a steady pace to avoid
185 double counting of birds. Individual transects took no more than 40 minutes to complete. All
186 transects were completed by TED and four local assistants throughout the survey period. Prior to
187 data collection, a significant training period (three months) in call identification was undertaken. This

188 included listening to recordings from Xeno-canto (www.xeno-canto.org), time in the field searching
189 for calling birds to visually determine their identity, and speaking to local people about the birds and
190 their calls. This experience meant identification of species from calls alone was straightforward
191 during the surveys. Distance estimation training was also undertaken, including estimating distances
192 to calling birds and before verifying the actual distance by pacing. This approach has been shown to
193 considerably improve the accuracy of distance estimates (Reynolds *et al.* 1980). Only one transect
194 was completed per day. Transects were typically repeated once per fortnight and each was surveyed
195 at least monthly. Each transect was repeated a minimum of six times, with at least 13 surveys
196 conducted in each habitat type. Transects within the same land-use type were not surveyed in the
197 same fortnight period and were as far apart from each other as possible given the mosaic of land
198 use, very steep terrain, and negotiated access with local landholders (transect centroids were
199 between 510 m and 2.5 km apart). The use of remote sensing data to measure proximity to other
200 habitats would have been of considerable value. However, high altitude areas, high humidity and
201 year-round abundant rainfall result in persistent cloud contamination, mean unobstructed very high
202 resolution images and LIDAR technology (see Goetz *et al.* 2009) to our knowledge, are not available
203 in the public domain for Kahua (see Garonna *et al.* 2009).

204 Prior to analysis, we excluded species that were in-flight and not directly interacting with the
205 habitat. The sole exception to this was the Pacific swallow (*Hirundo tahitica subfusca*) which forages
206 just above the uppermost vegetation strata and thus targets flying insects directly associated with
207 the habitat of interest. Swiftlets (*Collocalia esculenta*, *Aerodramus spodiopygius*, *A. v. vanikorensis*)
208 were excluded as they forage much higher above the uppermost vegetation strata and their
209 presence is more likely to be driven by broader-scale vegetation structure than the scale of the
210 transect.

211 In most bird surveys, not all species that are actually present are recorded (see Nichols and
212 Conroy 1996). The detectability of birds is typically lower in densely vegetated forests with tall
213 inaccessible canopies, such as intact forest, compared to more open modified habitats with broken,

214 lower canopies, such as monoculture cocoa (Gardner *et al.* 2009). Data were collected in a format
215 suitable for density estimation controlling for detectability variation using the standard Distance
216 methodology (Thomas *et al.* 2010). As accurate detectability functions require approximately 60
217 records per species per detectability class (in our case the five habitat classes), very few species were
218 recorded with sufficient frequency to reach this target. We therefore adopted an approach to
219 minimise the effects of detectability and to establish whether detectability effects may have biased
220 (or confounded) raw count comparisons across the habitats. Notably, we are interested in
221 determining which habitats have the highest relative species richness and abundance, rather than
222 precisely measuring abundance in each habitat, therefore if detectability of a species does not vary
223 across habitats then comparisons of raw counts across habitats are reasonably indicative of relative
224 abundance across habitats.

225 First, we minimised the effects of differences in detectability within our data by determining
226 the maximum appropriate transect width as 50 m on either side of the transect line (99% of all
227 records were obtained within these bounds) and excluded the small number of more distant
228 records. Second we established how detectability varied across habitat. For all species with >12
229 records we assessed the proportion of detections within 10 m in each habitat (versus 11-20, 21-30,
230 31-50 m distance bands) and applied a Kruskal-Wallis one-way ANOVA. We ran this test twice – first
231 across all habitats, and then across all habitats excluding intact forest to determine how different
232 intact forest was compared to other habitats, because we expected detectability to be lowest in this
233 habitat. Where significant differences were found, we examined each species and whether it was
234 more detectable in intact forest or more open habitats to establish any bias and to avoid erroneous
235 conclusions. For example, if a species was recorded more often in forest despite having lower
236 detectability there this would indicate robustly that the species was indeed more abundant in the
237 forest.

238

239 *Data analysis*

240 All analyses were conducted using R (version 3.0.1; R Core Team 2013) and focussed on presence-
241 absence and total count data at the habitat level (i.e. data was pooled across transects). We
242 calculated observed species richness (the total number of species recorded) for each habitat type.
243 We used EstimateS v9.0 (Colwell 2013) to calculate first-order jackknife, to estimate the total species
244 richness of each land use, and built sample based rarefaction curves to compare species richness
245 across land uses (Gotelli and Colwell 2001). We collated information and assigned each species a
246 category for endemism (Makira, Melanesia (using the biogeographical definition of Island
247 Melanesia), or widely distributed) following Dutson (2011) and functional traits (microhabitat
248 selection: water, aerial, canopy, canopy/understory, understory, or terrestrial; and foraging guild:
249 fruit and other vegetative material, nectarivore, omnivore, insectivore or vertebrates) following
250 Kratter *et al.* (2001) and our own extensive field experience (Appendix 1). We used non-parametric
251 tests to estimate the differences across land uses for species richness and endemism (Kruskal-
252 Wallis), and functional traits (Chi-squared).

253 First, we explored distribution between habitats by examining the presence or absence of a
254 species per 100 m section of each transect. We adopted this approach to minimize the effect of false
255 negatives associated with detectability and to allow a robust statistical evaluation controlling for
256 repeated measures within transects. We used the lme4 package in R. Binomial logistic regression
257 models with a log link function were constructed for each species with >12 records to predict
258 presence/absence per 100 m section with respect to habitat (categorical: intact forest as intercept,
259 secondary, garden, mixed-cocoa, monoculture) and controlling for time of day of observation (as a
260 covariate) and transect as a random effect. Probabilities of occurrence per 100 m of habitat were
261 then calculated by back-transforming the parameter estimates from each model and adding the
262 predicted value for the median time of day that the observations were carried out for each species
263 separately. We then assessed whether any of the differences in probability of occurrence were likely
264 to be driven by detectability in two stages. First if detectability (see above) did not vary across
265 habitats we assumed it was not a confounding effect for the species. Second if detectability did vary

266 across habitats we compared the rank order of occurrence with the pattern of detectability; if
267 species occurred more in habitats where they were less detectable we considered the occurrence
268 pattern robust and if the reverse occurred we considered that detectability was a potential
269 confound driving our occurrence pattern.

270 Very few species were identified as having their occurrence pattern confounded by
271 detectability (see below), and transect contributed very small and non-significant variation to the
272 occurrence models described above. Therefore we repeated the analysis using raw abundance data
273 from each 100 m of transect, ignoring detectability and considering each transect as independent, so
274 that we could maximize the data (i.e. variation in abundance rather than simply presence/absence)
275 within a relative simple modelling framework that would allow model convergence. We modelled
276 the difference in abundance of each species individually and from this inferred differences in
277 community assemblage between habitat types, using a multivariate generalised linear model on
278 species count data with the mvabund package (Wang *et al.* 2012). Multivariate generalised linear
279 models provide a powerful framework for analysing species abundance data and have been shown
280 to be more robust than distance based methods, such as multidimensional scaling and redundancy
281 analysis (Warton *et al.* 2012). The key advantage in this approach is that it has greater power to
282 detect patterns when analysing all species simultaneously. We included habitat (categorical: intact
283 forest, secondary, garden, mixed-cocoa, monoculture) and survey date as predictor variables. Date
284 was included to explore the possibility of seasonal differences. However, as we found no support for
285 seasonal effects ($p = 0.19$), it was removed from the model (Crawley 2007). A negative binomial
286 error structure was specified in our model, because our count data was over-dispersed (O'Hara and
287 Kotze 2010; Wang *et al.* 2012). Note though that this second modelling approach is simply another
288 way of testing for the effects of habitat type on individual species' density, and the results for each
289 species were very similar to the first presence/absence mixed model approach. We present the
290 simpler model approach in the main body of the paper and include results from the second
291 modelling approach as Appendix 2.

292

293 **Results**

294 We recorded a total of 3601 individuals from 42 bird species, including 12 of the 18 (67%) extant
295 endemic species on Makira (Appendix 1). The Shade Bush Warbler (*Cettia parens*), Island Leaf
296 Warbler (*Phylloscopus poliocephalus makirensis*), and Grey-throated White-eye (*Zosterops ugiensis*
297 *ugiensis*), are each known to occupy higher elevations (>500 m) than sampled in our study. We also
298 had no records of the Dusky Fantail (*Rhipidura tenebrous*) which is considered uncommon, the
299 nocturnal Makira Boobook (*Ninox roseoaxillaris*) which is considered rare (Dutson 2011), nor Makira
300 Moorhen (*Gallinula silvestris*) which is listed as Critically Endangered (BirdLife International 2013).

301

302 *Species richness*

303 The observed number of species found in each habitat ranged from 26 species in monoculture cocoa
304 to 34 species in secondary forest (Table 1). Rarefaction curves revealed that species richness had not
305 yet completely plateaued in any of the habitats, and therefore Jackknife species richness estimators
306 were higher than the observed species richness. This is the case for many surveys in tropical habitats
307 (Gotelli and Colwell 2001). Our observed species richness can therefore be considered an
308 underestimate of the actual number of species present. The mean number of species recorded per
309 transect differed across habitats ($H = 19.5$, $df = 4$, $p < 0.01$), and declined with increasing land use
310 intensity, with a mean of $15 (\pm 2.1)$ species in intact forest compared to a mean of $11 (\pm 2.3)$ species
311 recorded in monoculture cocoa (Table 1).

312

313 *Detectability*

314 Four species (out of 33) were found to have significant differences in their proportions of detection
315 within 10 m across habitats, these were: Rufous Fantail (*Rhipidura rufifrons russata*; $H = 18.7$, $df = 4$,
316 $p < 0.01$), Oriole Whistler (*Pachycephala orioloides christophori*; $H = 67.4$, $df = 4$, $p < 0.01$), Sooty
317 Myzomela (*Myzomela tristrami*; $H = 15.4$, $df = 4$, $p < 0.01$) and Yellow-bibbed Fruit Dove (*Ptilinopus*

318 *solomonensis ocularis*; $H=13.0$, $df=4$, $p<0.05$; Appendix 2). Species where detectability might have
319 confounded comparisons of abundance across habitats are highlighted in Appendix 2. If a species
320 was less detectable in intact forest and was also less commonly recorded in this habitat then its low
321 occurrence could be simply due to low detectability in intact forest. However, if a species was less
322 detectable in intact forest but was also recorded most commonly in this habitat then detectability
323 was not considered to be confounding. The Yellow-bibbed Fruit Dove and Mackinlay's Cuckoo-dove
324 (*Macropygia m. mackinlayi*) were both found to have higher detectability in intact forest and these
325 two species were not considered to be confounded by detectability in other more open habitats. The
326 Rufous Fantail and Sooty Myzomela were also both common in more open habitats, but were less
327 detectable in forest: therefore these two species may have been more common in forest than
328 recorded, although the result that both species were still abundant in open habitats remains. In
329 addition, Stephan's Emerald Dove (*Chalcophaps s. stephani*), Pacific Baza (*Aviceda subcristata*
330 *bismarckii*) and the Pacific Swallow were found to potentially be confounded, although these species
331 were only detected in low numbers in all habitats.

332

333 *Community assemblage*

334 A total of eleven endemic species were recorded in intact forest, as well as in secondary and mixed
335 cocoa habitats, with the lowest number of endemics (5) found in monoculture cocoa. Endemic
336 species were overlapping, with only the Makira Starling recorded in intact forest and not in any
337 other habitat, and the White-headed Fruit Dove recorded in secondary, garden and mixed-cocoa
338 (but not intact forest, or monoculture cocoa). The mean number of endemic species recorded per
339 transect also showed the same downward trend with increasing intensity of land use, with a mean of
340 $7 (\pm 1.6)$ endemics recorded per transect in intact forest compared to $3 (\pm 0.7)$ in monoculture cocoa
341 ($H = 30.9$, $df = 4$, $p<0.01$, Table 1, Figure 2). The total number of Melanesia-restricted range species
342 by habitat increased slightly with decreasing levels of land use intensification: 10 species in intact
343 forest, 11 in secondary forest, 11 in garden, 9 in mixed cocoa, and 8 in monoculture cocoa. However,

344 overall the mean number of Melanesia-restricted range species recorded per transect declined with
345 increasing intensity of land use from 6 (± 1.0) in intact forest to 2 (± 1.4) in monoculture cocoa, with
346 the differences across habitats found to be significant ($H = 30.7$, $df = 4$, $p < 0.01$, Table 1).

347 The relative distribution of functional traits displayed by birds differed between habitats,
348 both in terms of microhabitat selection ($\chi^2 = 141.2$, $df = 20$, $p < 0.01$) and foraging guild ($\chi^2 = 235.1$, df
349 $= 16$, $p < 0.01$). Monoculture cocoa had the highest percentage of canopy species (72%), followed by
350 mixed cocoa (64%) and garden (59%). Intact forest contained the highest proportion of
351 canopy/understory species (43%) followed by garden (39%). The highest proportion of terrestrial
352 species were recorded in monoculture cocoa (4%) and mixed cocoa (3%), with the highest
353 proportion of understory species recorded in forest areas (1%). Secondary forest contained the
354 highest percentage of frugivores (57%), followed by garden (45%) and mixed cocoa (38%). Garden
355 areas contained the highest percentage of insectivores (41%), followed by intact forest (39%). Intact
356 forest areas contained the highest percentage assemblage of omnivorous birds (9%) followed by
357 garden (4%). The highest percentage assemblage of nectarivorous birds was found in monoculture
358 cocoa (22%) followed by mixed cocoa (16%) and intact forest (15%).

359

360 *Species abundance*

361 We found significant differences in species abundance between habitats ($Dev = 800.2$, $df = 71$,
362 $p = 0.001$). The mvabund models suggested that 15 species had a distinct preference for one or more
363 habitats (Table 2, Figure 3). The presence/absence mixed models supported this finding, with 14 of
364 these 15 species found to have a distinct preference for one or more habitats (Appendix 2). The
365 additional species identified by the mvabund analysis was the Cardinal Lory (*Chalcopsitta cardinalis*):
366 sample sizes were at the low end for this species but 10/15 observations were from intact forest, the
367 habitat with the lowest detectability on average, so we categorised this species as being more
368 abundant in intact forest. Of the species that varied in abundance across habitats, 80 % were
369 restricted range species (Makira endemics, $n = 6$, and Melanesia endemics, $n = 6$). Seven of the

370 fifteen species found to vary by habitat were absent from the most heavily disturbed habitat. An
371 assessment of the mean number of these species seen per transect indicates that some species are
372 habitat specialists. For example Barred Cuckooshrike (*Coracina lineata makirae*), Cardinal Lory,
373 Makira Cicadabird (*C. salomonis*), Makira Honeyeater (*Meliarchus sclateri*), Oriole Whistler and
374 Yellow-bibbed Lory (*Lorius chlorocercus*) can be considered forest specialists with only limited
375 intrusion into other habitats (Figure 2). Conversely, Coconut Lorikeet (*Trichoglossus haematodus*
376 *massena*) and Collared Kingfisher (*Todiramphus chloris*) appear to benefit from agricultural
377 intensification because they only occupied modified habitats. In addition, species such as Chestnut-
378 bellied Monarch (*Monarcha castaneiventris megarhynchus*), Mackinlay's Cuckoo-dove (*Macropygia*
379 *m. mackinlayi*) and White-headed Fruit-dove (*Ptilinopus eugeniae*) appear to preferentially target
380 habitats with intermediate levels of disturbance (Figure 3).

381

382 **Discussion**

383 We assessed the impact of land-use change on the avifauna of the tropical island of Makira and
384 found species assemblage varied with land use. Endemic and restricted range bird species appear to
385 be particularly susceptible to increasing intensity of land use as many of these species are forest
386 specialists. As tropical islands have less functional redundancy than continental areas (McConkey
387 and Drake 2006), protecting the remaining forest will be critical to not only conserve these endemic
388 species, but also to maintain ecosystem functioning.

389

390 *Changes in species richness*

391 We found species richness to be similar across habitats, but that endemic species richness decreased
392 in habitats characterised by more intensive land uses. A similar trend has been observed in
393 Cameroon, where forest-dependent species were replaced by species typical of open habitats,
394 resulting in stable species richness but increasingly homogenous assemblages across habitats
395 (Lawton *et al.* 1998). Although species richness is a commonly used metric of biodiversity, it provides

396 only a coarse index with no information on abundance, or differentiation between vulnerable and
397 disturbance-adapted species (Pardini *et al.* 2009). Our findings that endemic species richness
398 declines with increasing intensity of land use is consistent with other studies (de Lima *et al.* 2012;
399 McKinney and Lockwood 1999). Furthermore, it supports the use of endemic species richness as a
400 useful metric to assess the conservation value of modified habitats (Waltert *et al.* 2011).

401

402 *Changes in species assemblage*

403 We found significant differences in the relative distribution of functional traits displayed by birds
404 between habitats, including a proportional increase in nectarivores and decrease in insectivores in
405 the most disturbed habitat, monoculture cocoa. Similar responses by nectarivores have been found
406 across similar land-use changes across the wider tropics (Şekercioğlu 2012) and is often attributed to
407 an increase in food resources (Waltert *et al.* 2005), which in Kahua is the presence of coconut and
408 ginger species. We also observed a proportional increase in frugivores in intermediately disturbed
409 areas of secondary and garden habitats. In contrast, insectivores and frugivores have been found to
410 consistently decline as intensity of land use increases (Marsden and Symes 2008; Newbold *et al.*
411 2013). Elsewhere this has been attributed to a reduced abundance in primary food types (Cleary *et*
412 *al.* 2007; Vetter *et al.* 2011). That the traditional agroforestry system in the Solomon Islands (and
413 across the wider Pacific region) generally preserves fruiting trees throughout the landscape (Mertz *et*
414 *al.* 2012) may explain our contrasting result, because even isolated fruiting trees in agricultural areas
415 have been shown to provide important resources for frugivorous birds (Şekercioğlu *et al.* 2007). The
416 change in feeding guilds we observed suggests bottom-up restrictions such as habitat and resource
417 limitations that likely included reductions in food availability.

418

419 *Changes in species abundance*

420 We found significant differences in species assemblages, with a suite of species showing distinct
421 habitat preferences. Such changes in abundance are known to have important effects on the

422 structure of communities and on the functioning of ecosystems (Gaston and Fuller 2008). A high
423 proportion of the species that were found to differ across habitats were endemic and Melanesia
424 restricted range species. The endemic Barred Cuckoo-shrike, Makira Honeyeater, Makira Cicadabird,
425 and restricted range Cardinal Lory and Yellow-bibbed Lory appear to be forest specialists with only
426 limited intrusion into other habitats. Stephan's Emerald Dove, which is a ground dove and
427 notoriously difficult to detect in dense forest environments may also be a forest specialist. The
428 Makira Cicadabird was not found in monoculture cocoa, and was also found to be less detectable in
429 intact forest suggesting that this species is also a forest specialist. These species are likely to be
430 sensitive to increasing intensity of land use. In contrast, the Oriole Whistler is a forest species that
431 appears to be able to occupy modified habitats, although the extent to which it is able to complete
432 its life cycle in more modified habitats is unknown. Similarly, although the abundance of Rufous
433 Fantail and Sooty Myzomela is likely to be higher in intact forest than detected in this study, both
434 species were also found to be common in degraded habitats, suggesting that a loss of forest may not
435 have such a strong impact on these species compared to more forest-specialist species. The White-
436 collared Monarch and Chestnut-bellied Monarch appear to be disturbance specialists because they
437 were found to occur at higher densities in areas of intermediate disturbance. This may be because of
438 increased feeding opportunities in lightly-disturbed environments, relative to heavily-disturbed
439 environments where a decrease in plant diversity is tracked by a corresponding decrease in
440 invertebrate diversity (Bennett and Gratton 2013).

441 We found two species that displayed an apparent preference for heavily disturbed
442 environments. The Collared Kingfisher and Coconut Lorikeet were recorded in all habitats, but
443 displayed a trend towards higher numbers in heavily disturbed habitats especially cocoa
444 monocultures. Consistent with their ability to exploit human modified habitats both of these species
445 are of lower conservation concern (IUCN 2014).

446

447 *Implications of future land use change*

448 Our findings are consistent with other studies that suggest replacing forests with more intensive
449 land uses results in a shift toward bird communities with less specialized traits and a reduction in the
450 diversity for some key functional groups (Luck *et al.* 2013; Şekerciöğlü 2012). Land-use change
451 presents a serious threat to forest birds, especially for those with specialised functional traits
452 (Newbold *et al.* 2013). This trend is a particular concern for tropical islands because not only do
453 islands have endemic fauna more prone to extinction (88% of bird extinctions since 1600 have
454 occurred on islands, Butchart *et al.* 2006), they also have less functional redundancy (McConkey and
455 Drake 2006; Wardle 2002). Thus, the loss of island species that fill key functional roles may
456 inordinately disrupt vital ecosystem services and processes, making ecosystems less resilient to
457 environmental change (Hooper *et al.* 2005). In areas where the human population relies heavily on
458 forest products, as is the case on Makira, such disruption might increase social and economic
459 vulnerability, and ultimately contribute to the degradation of socio-ecological systems (Elmqvist *et*
460 *al.* 2003).

461 Shifting cultivation has occurred on the Solomon Islands for at least 3000 years (Mertz *et al.*
462 2012) and consequently it is likely that the most sensitive species became extinct a long time ago
463 (e.g. large flightless birds that were easy to catch with dogs). Therefore, the remaining fauna consists
464 of species that are already adapted to some level of human disturbance; from low disturbance intact
465 forest to high disturbance plantations across the broader landscape mosaic. However, the recent
466 introduction of cash crops and a growing human population has promoted a shift towards more
467 intensive land uses. Avifauna assemblages appear most negatively affected by this intensive land
468 use, with many species of conservation concern not found in this habitat. Currently, these
469 monocultures are spatially constrained and cover relatively small areas in Kahua, but should their
470 development continue on the current trajectory, negative repercussions for biodiversity and
471 ecosystem function would appear inevitable.

472

473 *Implications for conservation*

474 Despite exceptional levels of endemism on Makira and across the Solomon Islands, relatively little is
475 known about the distribution, ecology and threats affecting many endemic taxa. This is an urgent
476 requirement as many of the endemic species detected in our study appear to be specialists of intact
477 forest and it is these species that experience higher rates of extinction (Ricklefs and Bermingham
478 2002). Thus, an important management step will be to conserve the extent and quality of remaining
479 intact forest. Currently the Solomon Islands has ineffective environmental legislation and policy
480 (UNDP 2006) that could be more appropriately aligned with biodiversity value, to better represent
481 naturally occurring habitats from the coast to higher elevations as has been recommended
482 elsewhere (e.g. Mallari *et al.* 2011). Ensuring the protection of intact forest, particularly in areas
483 below 400m, where logging is considered commercially more desirable, will be important, as these
484 areas are currently unprotected and often under the greatest pressure from logging and plantation
485 development (Buchanan *et al.* 2008).

486

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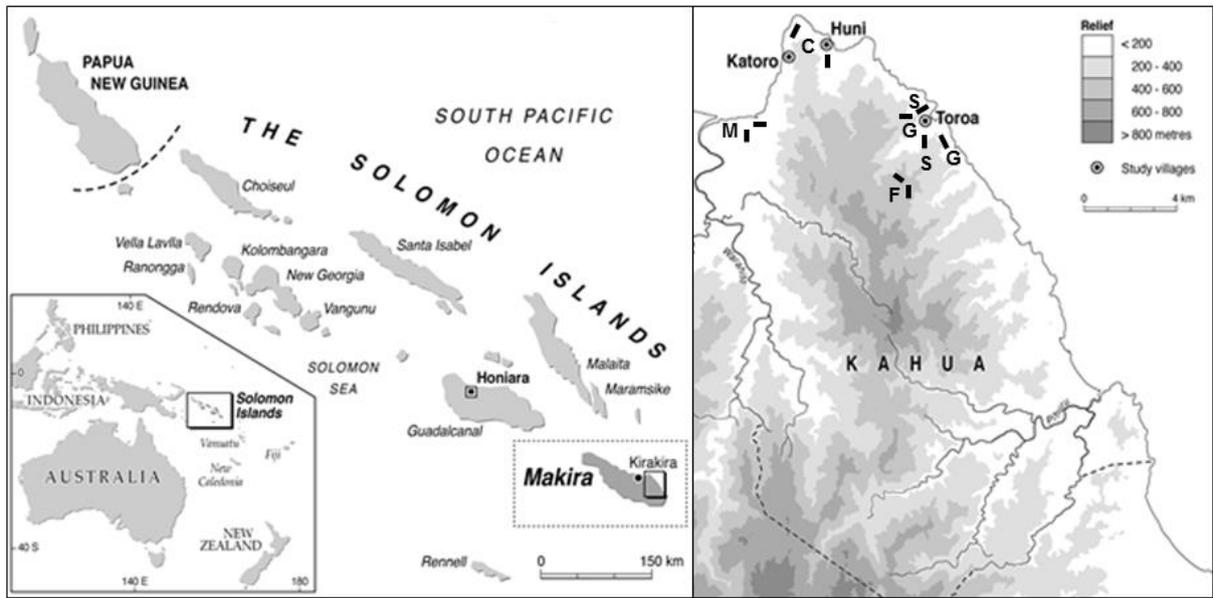
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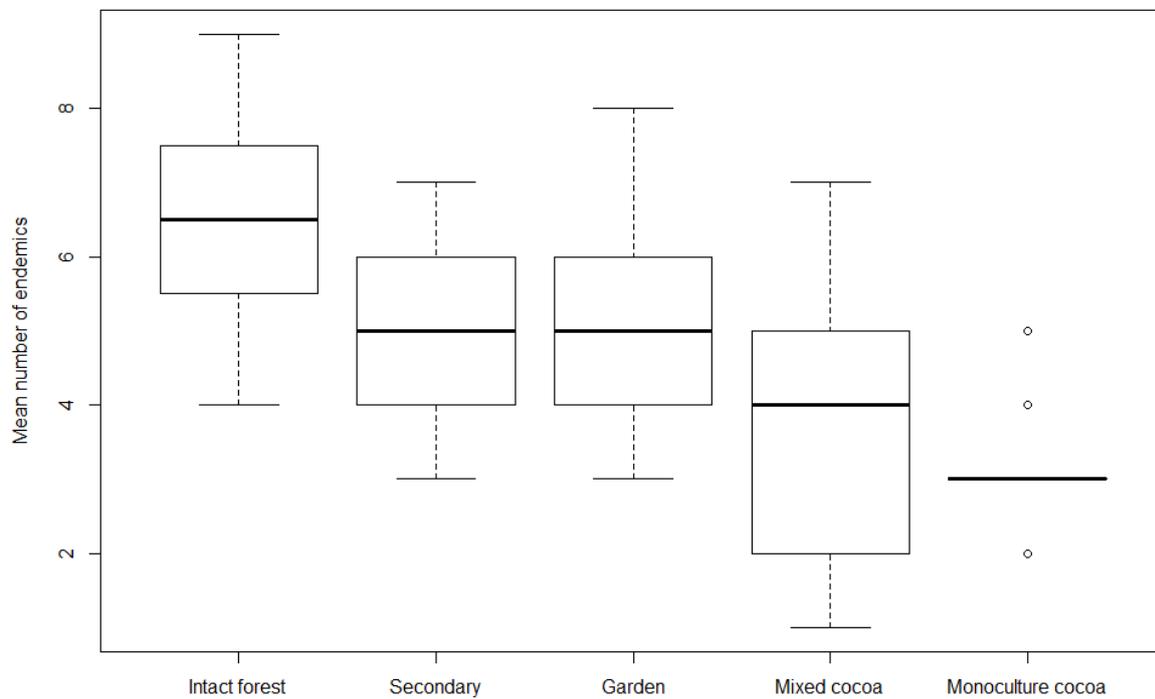
772 **Figure 1.** Location of Kahua study site, and location of the ten transects in the different habitats (F=
 773 forest, S= secondary, G= garden, C= mixed cocoa, M = monoculture cocoa).



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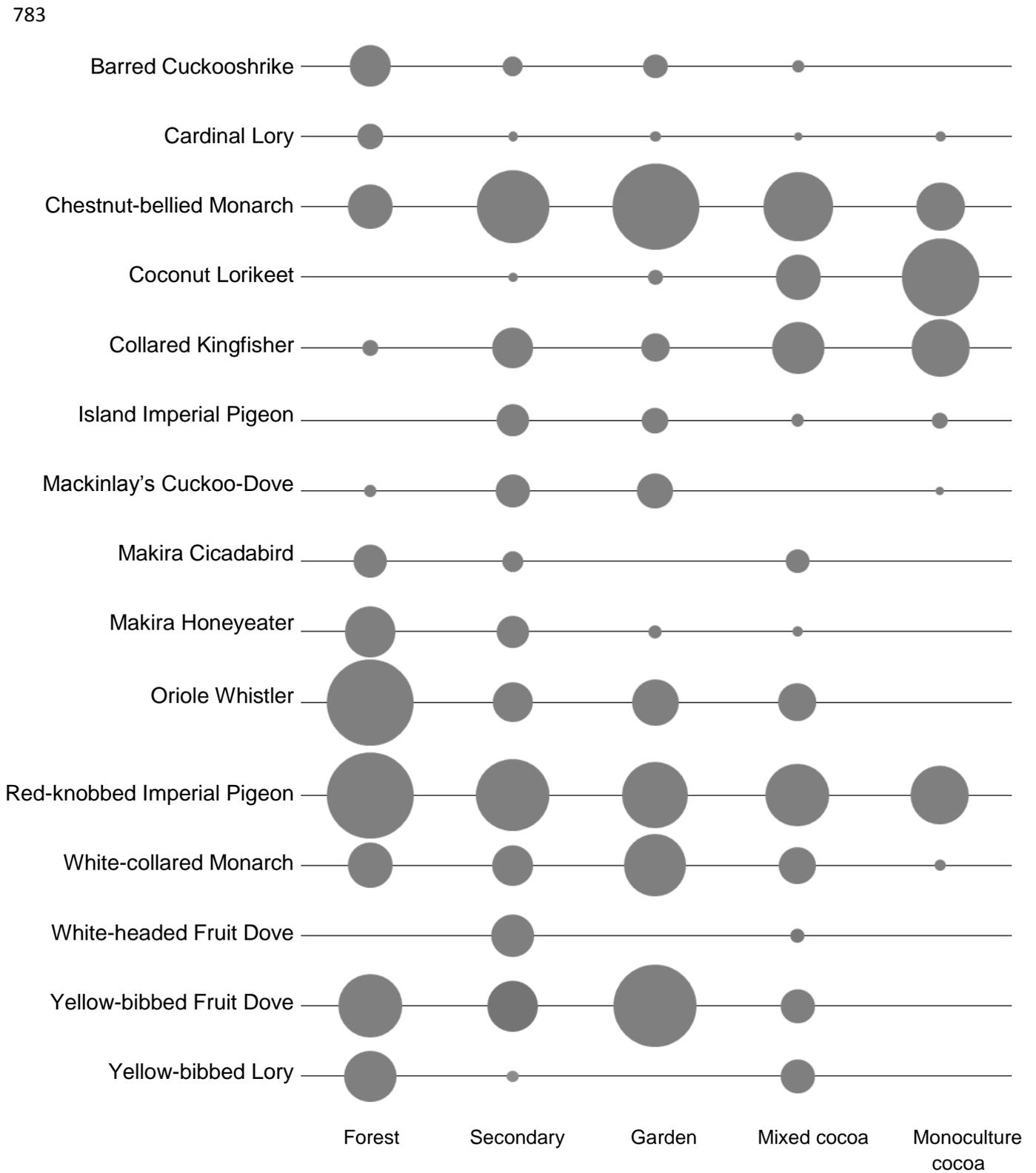
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776 **Figure 2.** Box plots providing median, quartiles & 95% confidence interval of Makira endemics per
777 transect for the five land use types sampled.



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780 **Figure 3.** Relative abundance across habitats for species which showed significant differences in
 781 their distribution. Abundance corresponds to the size of the circle (Raw abundance values shown in
 782 Appendix 2).



784 **Table 1.** Summary statistics for avian species richness metrics within the five dominant habitats on
 785 Makira, Solomon Islands.

Habitat	Species richness per transect	Mean (±SD)	Total number of endemic sp.	Mean number of endemic sp. per transect (±SD)	Total number of restricted sp.	Mean number of restricted sp. per transect (±SD)
Forest	27	15 (±2.1)	11	7 (±1.6)	10	6 (±1.0)
Secondary	34	14 (±3.4)	11	5 (±1.2)	11	5 (±1.6)
Garden	30	13 (±3.8)	9	5 (±1.5)	11	5 (±1.7)
Mixed cocoa	33	12 (±1.8)	11	4 (±1.7)	9	4 (±1.6)
Monoculture	26	11 (±2.3)	5	3 (±0.7)	8	2 (±1.4)

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788 **Table 2.** Variation of avian species assemblage between habitats on Makira, Solomon Islands
789 (mvabund: Dev = 800.2, d.f. = 71, $P = 0.001$). Scientific names of all species are given in Appendix 1.
790 Dev, Deviance; P-value; N, number of individuals recorded; Variation indicates species that varied in
791 abundance between habitats, with final column listing the habitat in which they were found to be
792 most abundant. Species are presented, first by those that varied between habitats from intact forest
793 to monoculture cocoa, and secondly by Family. The remaining species that did not vary between
794 habitats are ordered by Family.

Species	Dev	P-value	N	Variation	Habitat in which most abundant
Barred Cuckoo-shrike	22.1	0.004	36	x	Intact forest
Makira Cicadabird	25.1	0.002	55	x	Intact forest
Red-knobbed Imperial-Pigeon	25.5	0.002	222	x	Intact forest
Makira Honeyeater	83.7	0.001	70	x	Intact forest
Yellow-throated Whistler	87.6	0.001	228	x	Intact forest
Cardinal Lory	18.7	0.024	15	x	Intact forest
Yellow-bibbed Lory	32.1	0.001	43	x	Intact forest
Yellow-bibbed Fruit-Dove	70.3	0.001	124	x	Intact forest, Garden
Island Imperial-Pigeon	20.9	0.010	29	x	Secondary forest
Mackinlay's Cuckoo-Dove	36.1	0.001	35	x	Secondary forest
White-headed Fruit-Dove	22.8	0.002	33	x	Garden
White-collared Monarch	24.3	0.002	77	x	Garden
Chestnut-bellied Monarch	26.4	0.001	236	x	Garden
Collared Kingfisher	28.1	0.001	100	x	Monoculture Cocoa
Coconut Lorikeet	36.8	0.001	94	x	Monoculture Cocoa
Pied Goshawk	12.8	0.19	12		
Pacific Baza	3.2	0.73	3		
Brahminy Kite	3.5	0.72	1		
Common Kingfisher	7.5	0.56	4		
Variable Dwarf Kingfisher	15.2	0.08	13		
Beach Kingfisher	8.0	0.53	3		
Nankeen Night-Heron	6.8	0.56	3		
Long-tailed Triller	6.9	0.56	58		
Stephan's Emerald Dove	6.2	0.56	14		
Chestnut-bellied Imperial-Pigeon	16.0	0.06	65		
Eastern Bronze Ground-Dove	7.8	0.56	5		
Crested Cuckoo-Dove	11.7	0.28	13		
Dollarbird	9.4	0.41	16		
Pacific Koel	9.2	0.41	4		
Mottled Flowerpecker	8.9	0.45	94		
Pacific Swallow	8.5	0.49	12		
Melanesian Megapode	11.0	0.29	16		
Cardinal Myzomela	14.3	0.11	75		
Sooty Myzomela	10.6	0.29	183		
Eclectus Parrot	1.2	0.89	39		

Pale-vented Bush-hen	9.4	0.41	3
Buff-banded Rail	4.0	0.72	2
Rufous Fantail	14.9	0.08	252
Willie Wagtail	13.6	0.14	19
Makira Starling	3.8	0.72	10
Metallic Starling	7.9	0.54	270
Singing Starling	7.6	0.56	146

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796

797 **Appendix 1.** Species guilds for all bird species recorded during surveys on Makira, Solomon Islands*.

798 Where relevant trinomials are provided.

English name	Scientific name	Endemic	Micro-habitat	Die t	IUC N
Pied Goshawk	<i>Accipiter albogularis albogularis</i>	Makira	C	V	NA
Pacific Baza	<i>Aviceda subcristata bismarckii</i>		C	V	LC
Solomons Sea-Eagle	<i>Haliaeetus sanfordi</i>	Melanesia	AE	V	VU
Brahminy Kite	<i>Haliastur indus flavirostris</i>	Melanesia	AE	V	LC
Common Kingfisher	<i>Alcedo atthis salomonensis</i>	Melanesia	AQ	V	LC
Variable Dwarf Kingfisher	<i>Ceyx lepidus gentianus</i>	Makira	U	V	NA
Collared Kingfisher	<i>Todiramphus chloris</i>		C	V	LC
Beach Kingfisher	<i>Todiramphus saurophagus admiralitatis</i>		AQ	V	LC
White-rumped Swiftlet	<i>Aerodramus spodiopygius</i>		AE	I	LC
Uniform Swiftlet	<i>Aerodramus vanikorensis vanikorensis</i>		AE	I	LC
Glossy Swiftlet	<i>Collocalia esculenta</i>		AE	I	LC
Nankeen Night-Heron	<i>Nycticorax caledonicus mandibularis</i>		AQ	FI	LC
Barred Cuckoo-shrike	<i>Coracina lineata makirae</i>	Makira	C	O	LC
Makira Cicadabird	<i>Coracina salomonis</i>	Makira	C	O	NA
Long-tailed Triller	<i>Lalage leucopyga affinis</i>	Melanesia	C	I	LC
Stephan's Emerald Dove	<i>Chalcophaps stephani stephani</i>		T	FR	LC
Chestnut-bellied Imperial-Pigeon	<i>Ducula brenchleyi</i>	Melanesia	C	FR	VU
Island Imperial-Pigeon	<i>Ducula pistrinaria pistrinaria</i>		C	FR	LC
Red-knobbed Imperial-Pigeon	<i>Ducula rubricera rubricera</i>	Melanesia	C	FR	NT
Eastern Bronze Ground-Dove	<i>Gallicolumba beccarii solomonensis</i>	Melanesia	T	FR	LC
Mackinlay's Cuckoo-Dove	<i>Macropygia mackinlayi mackinlayi</i>	Melanesia	C	FR	LC
White-headed Fruit-Dove	<i>Ptilinopus eugeniae</i>	Makira	C	FR	NT
Yellow-bibbed Fruit-Dove	<i>Ptilinopus solomonensis solomonensis</i>	Makira	C	FR	LC
Crested Cuckoo-Dove	<i>Reinwardtoena crassirostris</i>	Melanesia	C	FR	NT
Dollarbird	<i>Eurystomus orientalis crassirostris</i>		C	I	LC
Pacific Koel	<i>Eudynamis orientalis alberti</i>		C	O	LC
Mottled Flowerpecker	<i>Dicaeum tristrami</i>	Makira	CU	N	LC
Pacific Swallow	<i>Hirundo tahitica subfusca</i>		AE	I	LC
Melanesian Megapode	<i>Megapodius eremita</i>	Melanesia	T	O	LC
Makira Honeyeater	<i>Meliarchus sclateri</i>	Makira	C	N	NA

Cardinal Myzomela	<i>Myzomela cardinalis pulcherrima</i>		C	N	LC
Sooty Myzomela	<i>Myzomela tristrami</i>	Makira	C	N	LC
Chestnut-bellied Monarch	<i>Monarcha castaneiventris megarhynchus</i>	Melanesi a	CU	I	LC
White-collared Monarch	<i>Symposiachrus vidua vidua</i>	Makira	CU	I	NA
Yellow-throated Whistler	<i>Pachycephala orioloides christophori</i>	Melanesi a	C	O	NA
Cardinal Lory	<i>Chalcopsitta cardinalis</i>	Melanesi a	C	FR	LC
Eclectus Parrot	<i>Eclectus roratus</i>		C	FR	LC
Yellow-bibbed Lory	<i>Lorius chlorocercus</i>	Melanesi a	C	FR	LC
Coconut Lorikeet	<i>Trichoglossus haematodus massena</i>		C	FR	LC
Pale-vented Bush-hen	<i>Amaurornis moluccana ultima</i>		T	O	LC
Buff-banded Rail	<i>Gallirallus philippensis meyeri</i>	Melanesi a	T	O	NA
Willie Wagtail	<i>Rhipidura leucophrys</i>		T	I	LC
Rufous Fantail	<i>Rhipidura rufifrons russata</i>	Makira	CU	I	NA
Singing Starling	<i>Aplonis cantoroides</i>		C	FR	LC
Makira Starling	<i>Aplonis dichroa</i>	Makira	C	FR	LC
Metallic Starling	<i>Aplonis metallica nitida</i>		C	FR	LC

799

800 **Microhabitat:** AQ = water and water edge; AE = aerial; C = canopy; CU= canopy/understory;

801 U = understory; T = terrestrial.

802 **Prey:** FR = fruit and other vegetative material; N = nectarivore; O = omnivore; I = insects; V =

803 vertebrates.

804 **IUCN Red List status:** VU=vulnerable; NT=near threatened; LC= least concern; NA = not yet assessed

805 for the IUCN Red List.

806 * Information collated from Dutson (2011), Kratter *et al.* (2001), IUCN (2014) and our own extensive

807 field experience.

808

809 **Appendix 2:** Variation in detectability and probability of occurrence of bird species in 100m sections
 810 of transects across habitats. Species are listed in order of abundance on transects. Forest, intact forest;
 811 Sec, secondary forest; Garden; Cocoa, mixed Cocoa; MCocoa, monoculture Cocoa.

812 The left-hand component of the table indicates the proportion of sightings of each species within 10 m
 813 of the transect line in each habitat (versus the other distance classes). A dash indicates the species was
 814 recorded opportunistically in the habitat but not during or according to the transect methodology. The
 815 habitat with the lowest detectability is highlighted in bold; a high proportional value indicates a
 816 greater proportion of sightings at relatively close range, indicating lower detectability. The results of a
 817 Kruskal–Wallis one-way ANOVA of detectability distances across all habitats (Forest column) and
 818 then across all habitats excluding intact forest (Sec column) are given below the proportions for each
 819 species (significant difference in detectability distance, $P > 0.05$), shown as: MORE, which indicates a
 820 species was significantly more detectable in open, less-forested habitats as expected (refer to the main
 821 text for further information on detectability); or LESS, which indicates a species was significantly
 822 more detectable in intact or secondary forest habitats; ns, non-significant ; SIG, significant, this refers
 823 to the second test only and indicates whether there was significant variation across all habitats
 824 excluding intact forest (Sec column)

825 The right-hand component of the table gives the predicted probability of occurrence of each species in
 826 each habitat. The predicted values are derived from a model predicting the presence or absence of a
 827 species within a 100-m transect section of a given habitat, with time of day as a confounding variable
 828 and transect included as a random effect. The predicted values for each habitat are given at the median
 829 time that the species was observed, back transformed into probabilities of occurrence per 100 m on a
 830 transect (i.e. a value of 0.20 indicates that the species was recorded on average once every 500 m).
 831 Intact forest was set as the reference category in the model (as the control, because it is the most
 832 natural habitat) and if there were significant differences in probability of occurrence in other habitats
 833 compared with intact forest these are indicated with the predicted value in bold. Total counts for each
 834 species (i.e. raw abundance data) are given below the predicted probabilities for each habitat. Note
 835 that there is not a perfect correspondence between raw abundance and predicted probabilities because
 836 the predicted probabilities are modelled from presence and absence, not abundance, and there was a
 837 strong effect of time of day (probability of detection decreasing with time) that was variable by
 838 species

839 Yes in the final column indicates significant differences in detectability, or potential differences in
 840 detectability that could not be measured (see explanation of left-hand component of table), that may
 841 have confounded comparisons of abundance across habitats; No indicates there were no significant
 842 differences or potential differences in detectability; species that had less than 12 records were not
 843 included in the analysis and cells have been left blank.

844

Species	Proportion detected <10m					Probability of occurrence & total count					Potential confound?
	Forest	Sec	Garden	Cocoa	MCocoa	Forest	Sec	Garden	Cocoa	MCocoa	
Red-knobbed Imperial Pigeon	0.37 NS	0.22 NS	0.29	0.31	0.32	0.55 87	0.41 60	0.30 44	0.33 63	0.24 28	No
Metallic Starling	- NS	0.22 NS	0.75	0.17	0.25	0.02 7	0.17 94	0.10 45	0.17 83	0.13 39	No
Rufous Fantail	0.85 MORE	0.58 SIG	0.69	0.51	0.47	0.36 42	0.51 38	0.71 65	0.52 62	0.53 45	Yes
Oriole Whistler	0.45 MORE	0.06 NS	0.09	0.14	-	0.75 164	0.18 22	0.32 27	0.1 27	0.00 0	No

Chestnut-bellied Monarch	0.58 NS	0.5 NS	0.62	0.48	0.52	0.24 30	0.57 52	0.6 65	0.47 64	0.28 25	No
Sooty Myzomela	1.0 MORE	0.89 SIG	0.75	0.75	0.71	0.09 15	0.43 52	0.27 23	0.37 49	0.41 44	Yes
Chestnut-bellied Imperial Pigeon	0.24 NS	0.36 NS	0.29	0.17	0.5	0.18 19	0.21 29	0.26 79	0.15 28	0.08 8	No
Singing Starling	- -	0.20 NS	-	0.00	0.00	0 0	0.07 110	0.01 13	0.06 11	0.02 12	No
Yellow-bibbed Fruit Dove	0.13 LESS	0.28 SIG	0.44	0.00	-	0.41 48	0.26 20	0.42 46	0.12 12	0.02 1	No
Coconut Lorikeet	- -	- NS	-	0.00	0.53	0 0	0.01 1	0.01 2	0.09 32	0.43 82	No
Eclactus Parrot	0.00 NS	0.50 NS	0.00	0.00	0.00	0.04 5	0.16 11	0.07 10	0.29 68	0.12 9	No
Collared Kingfisher	0.20 NS	0.23 NS	0.20	0.13	0.10	0.05 5	0.24 17	0.11 7	0.29 36	0.42 36	No
Mottled Flowerpecker	0.83 NS	0.71 NS	0.72	0.58	0.81	0.21 28	0.2 17	0.14 11	0.12 19	0.24 21	No
White-collared Monarch	0.63 NS	0.60 NS	0.75	0.67	1.00	0.17 23	0.11 13	0.22 25	0.04 14	0.01 1	No
Cardinal Myzomela	1.00 NS	0.89 NS	0.60	0.77	0.68	0.03 3	0.13 14	0.07 5	0.17 26	0.31 27	No
Makira Honeyeater	0.42 NS	0.36 NS	0.00	0.5	-	0.37 54	0.18 14	0.03 2	0.02 2	0.00 0	No
Variable Dwarf Kingfisher	1.00 NS	0.60 NS	1.00	0.40	0.72	0.02 2	0.06 8	0.06 6	0.06 24	0.09 16	Yes
Makira Cicadabird	0.41 MORE	0.2 NS	0.2	0.13	-	0.12 23	0.09 5	0.11 15	0.13 11	0.00 0	Yes
Yellow-bibbed Lory	0.25 NS	- -	-	0.00	-	0.15 31	0.00 0	0.01 1	0.09 18	0.00 0	No
Mackinlay's Cuckoo-Dove	0.5 LESS	0.56 NS	0.72	-	0.00	0.02 3	0.15 16	0.19 15	0.00 0	0.01 1	No
Barred Cuckooshrike	0.56 NS	1.00 NS	1.00	0.00	1.00	0.09 25	0.03 4	0.02 3	0.02 2	0.01 1	No
White-headed Fruit Dove	- -	0.67 NS	0.33	-	-	0.00 0	0.08 14	0.12 16	0.01 2	0.00 0	No
Island Imperial Pigeon	- -	0.25 NS	0.00	0.00	0.75	0.00 0	0.26 14	0.17 8	0.04 3	0.09 4	No
Willie Wagtail	- -	- NS	0.33	0.6	0.25	0.00 0	0.01 1	0.06 4	0.05 5	0.11 9	No
Long-tailed Triller	- -	- NS	0.5	0.2	0.5	0.00 0	0.01 1	0.04 3	0.05 6	0.06 6	No
Melanesian Megapode	0.00 NS	1.00 NS	-	0.25	1.00	0.07 6	0.04 2	0.00 0	0.05 7	0.02 1	No
Cardinal Lory	- -	- -	-	-	-	0.09 10	0.02 1	0.02 1	0.00 1	0.03 2	No
Stephan's Emerald Dove	1.00 NS	1.00 NS	-	-	0	0.01 1	0.07 6	0.00 0	0.03 4	0.04 3	Yes
Brahminy Kite	- -	- -	-	-	-	0 0	0.07 6	0.05 3	0.02 3	0.03 2	No
Crested Cuckoo-dove	0.00 NS	0.00 NS	0.00	0.00	0.00	0.02 2	0.07 6	0.06 4	0.01 1	0.00 0	No
Pacific Baza	- -	- -	-	-	-	0.01 1	0 0	0.03 2	0.02 5	0.03 4	Yes
Pied Goshawk	- NS	0.50 NS	-	0.50	-	0.02 2	0.03 3	0.00 0	0.05 7	0.00 0	No
Pacific Swallow	- -	- -	-	-	-	0.00 0	0.03 7	0.01 5	0.00 0	0.00 0	Yes
Bronze Ground Dove						0	3	0	0	2	

Pacific Koel	3	0	0	1	0
Common Kingfisher	0	0	0	1	3
Beach Kingfisher	0	0	0	0	3
Pale-vented bushhen	0	0	0	0	3
Nankeen Night Heron	0	3	0	0	0
Solomons Sea Eagle	1	0	0	2	0
Buff-banded Rail	0	1	1	0	0
Makira Starling	1	0	0	0	0

845