



Kent Academic Repository

Gardner, Charlie J., Jasper, Louise D., Eonintsoa, Christian, Duchene, Julio-Joseph and Davies, Zoe G. (2016) *The impact of natural resource use on bird and reptile communities within multiple-use protected areas: evidence from sub-arid southern Madagascar*. *Biodiversity and Conservation*, 25 . pp. 1773-1793. ISSN 0960-3115.

Downloaded from

<https://kar.kent.ac.uk/84417/> The University of Kent's Academic Repository KAR

The version of record is available from

<https://doi.org/10.1007/s10531-016-1160-4>

This document version

Author's Accepted Manuscript

DOI for this version

Licence for this version

UNSPECIFIED

Additional information

Versions of research works

Versions of Record

If this version is the version of record, it is the same as the published version available on the publisher's web site. Cite as the published version.

Author Accepted Manuscripts

If this document is identified as the Author Accepted Manuscript it is the version after peer review but before type setting, copy editing or publisher branding. Cite as Surname, Initial. (Year) 'Title of article'. To be published in *Title of Journal* , Volume and issue numbers [peer-reviewed accepted version]. Available at: DOI or URL (Accessed: date).

Enquiries

If you have questions about this document contact ResearchSupport@kent.ac.uk. Please include the URL of the record in KAR. If you believe that your, or a third party's rights have been compromised through this document please see our [Take Down policy](https://www.kent.ac.uk/guides/kar-the-kent-academic-repository#policies) (available from <https://www.kent.ac.uk/guides/kar-the-kent-academic-repository#policies>).

1 Original research

2

3 **The impact of natural resource use on bird and reptile**
4 **communities within multiple-use protected areas: evidence from**
5 **sub-arid southern Madagascar**

6

7 Charlie J. Gardner^{I, II*}, Louise D. Jasper^{III}, Christian Eonintsoa^{IV}, Julio-Josepha Duchene^{IV}
8 and Zoe G. Davies^{II}

9

10 ^I *WWF Madagascar and Western Indian Ocean Programme Office, BP 738, Antananarivo*
11 *101, Madagascar.*

12 ^{II} *Durrell Institute of Conservation and Ecology (DICE), School of Anthropology and*
13 *Conservation, University of Kent, Canterbury, Kent, CT2 7NR, UK*

14 ^{III} *BP 639, Toliara 601, Madagascar*

15 ^{IV} *Département de Biologie, Faculté des Sciences, Université de Toliara, BP 185, Toliara*
16 *601, Madagascar*

17 * Corresponding author: cg399@kent.ac.uk

18

19 Running title: *Degradation impacts in multiple-use protected areas*

20 Word count: 5776 (+ abstract 250, references 3269)

21 No. of figures: 2 (+ 2 in supplementary materials)

22 No. of tables: 5 (+ 4 in supplementary materials)

25 **Abstract**

26 Multiple-use protected areas, in which sustainable levels of extractive livelihood activities are
27 permitted, play an increasingly important role in the global protected area estate, and are
28 expected to rise in prevalence. However, we know little about their effectiveness at
29 conserving biodiversity. We surveyed bird and reptile communities in three areas across a
30 forest disturbance gradient resulting from charcoal production and shifting cultivation within
31 a multiple-use protected area in Madagascar's sub-arid spiny forest. We scored individual
32 species using a Conservation Value Index (CVI; a simple metric based on rarity, threat and
33 distinctiveness), and estimated the total conservation value of each treatment by calculating
34 the sum of frequency-weighted CVI scores across all present species. Bird and reptile
35 community responses to forest disturbance were idiosyncratic. Bird richness was greatest in
36 the moderate-disturbance treatment, but the low-disturbance treatment had the superior
37 conservation value due to higher frequencies of locally-endemic species. Reptile richness was
38 the same in low- and moderate-disturbance treatments, but the conservation value of the latter
39 was greater. The high-disturbance areas had lowest richness and conservation value for both
40 groups. For birds, increasing disturbance levels were accompanied by community turnover
41 from high-value to low-value species, a pattern highlighted by CVI that is masked by
42 assessing species richness alone. Although some endemic species appear to be resilient to
43 degradation, multiple-use protected areas in Madagascar may lose biodiversity since most
44 endemic species are forest-dependent. Stricter protected area models may be more
45 appropriate in areas where much of the high-value biodiversity is sensitive to habitat
46 degradation.

47

48 **Keywords:** Conservation value; Degradation; Dry forest; Faunal communities; Sustainable
49 Use

50

51

52 **1 Introduction**

53 The impacts of human activity now threaten most of the Earth's species and ecosystems
54 (Ehrlich and Pringle 2008) and have precipitated the planet's sixth mass extinction (Barnosky
55 et al. 2011). Our primary strategy to stem this biodiversity loss is the creation and
56 management of protected areas, which cover over 15 % of the world's land area and
57 constitute the largest planned land use in history (Juffe-Bignoli et al. 2014). All protected
58 areas are spaces "recognised, dedicated and managed... to achieve the long-term conservation
59 of nature with associated ecosystem services and cultural values" (Dudley 2008), but they
60 vary greatly in management objective and approach. These differences form the basis for the
61 World Conservation Union's (IUCN) protected area categorisation system (Dudley 2008;
62 Dudley et al. 2010). For simplicity's sake the categories are often divided into 'strict'
63 protected areas (generally categories I-IV), which seek to isolate nature from human
64 processes that threaten it, and 'multiple-use' sites, which promote conservation through the
65 sustainable extractive use of natural resources (category VI) or traditional land uses that
66 sustain biodiversity (category V).

67

68 Recent decades have seen the number of multiple-use protected areas grow significantly in
69 many parts of the world (Juffe-Bignoli et al. 2014). Although some strict sites have been
70 downgraded (Mascia et al. 2014), this has been driven primarily by the predominance of
71 multiple-use categories amongst new protected areas (Zimmerer et al. 2004). The trend can
72 largely be attributed to: i) the lack of remaining 'wilderness' areas, with a low human
73 footprint, suitable for the creation of strict categories (Leroux et al. 2010); and, ii) a paradigm
74 shift in conservation, reflecting concern for the effects of exclusionary approaches on human
75 wellbeing (Adams and Hutton 2007; Miller 2014), and the suggestion that sustainable use

76 may be a more effective long-term conservation strategy than strict protection (Rosser and
77 Leader-Williams 2010). As a result, only 45 % of the world’s protected areas are assigned to
78 categories I-IV (Jenkins and Joppa 2009), and category VI sites expanded from 14 to 32 % of
79 the world’s protected area estate (by area) between 1990 and 2010 (Bertzky et al. 2012). This
80 trend is expected to become even more pronounced in the future (McDonald and Boucher
81 2011).

82

83 Signatories to the Convention of Biological Diversity are expected to increase the coverage
84 of terrestrial protected areas to 17 % of their national territory by 2020 and ensure that they
85 are “effectively managed” (CBD 2010, Aichi Target 11), a target that will require the most
86 rapid expansion of protected areas in history (Venter et al. 2014). Thus, if new protected
87 areas are expected to largely comprise multiple-use categories, it is important to know
88 whether they are likely to be successful at achieving their objective – the long-term
89 conservation of nature – in the face of authorised human impacts (Dudley et al. 2014; Watson
90 et al. 2016). This is particularly apposite given longstanding debates over the contribution of
91 multiple-use protected areas to conservation goals (Locke and Dearden 2005; Gaston et al.
92 2008; Shafer 2015).

93

94 The effectiveness of protected areas depends on both their coverage (i.e. ensuring that
95 maximum biodiversity is represented within them) and their success in buffering the
96 biodiversity from the processes that threaten its viability (Gaston et al. 2008; Watson et al.
97 2014). However, research tends to concentrate on the former (e.g. Montesino Pouzols et al.
98 2014; Venter et al. 2014; Butchart et al. 2015; Polak et al. 2015; Visconti et al. 2015), with
99 the result that we know little about the success of protected areas in maintaining their
100 condition over time (Cabeza 2013; Geldmann et al. 2013; Beaudrot et al. 2016; Watson et al.

101 2016). This knowledge gap is particularly acute with regards to multiple-use categories.
102 Global studies comparing across categories have found stricter protected areas to be more
103 effective at slowing deforestation in some regions (Joppa and Pfaff 2011; Scharlemann et al.
104 2010), whereas multiple-use sites demonstrate greater success in other countries (Ferraro et
105 al. 2013; Nelson and Chomitz 2011). However, the use of remote sensed data within such
106 analyses only allows us to quantify vegetation cover, therefore providing little insight into the
107 ecological integrity of remaining natural vegetation and faunal communities beneath the
108 canopy (Peres et al. 2006; Beaudrot et al. 2016). Less conspicuous changes to forest structure
109 and composition (i.e. forest degradation) can stem from activities such as non-industrial
110 selective logging, fuelwood collection, livestock grazing and the harvesting of non-timber
111 forest products (NTFPs). Typically, these are precisely the types of activity that may be
112 sanctioned within category V and VI protected areas (Dudley 2008). Indeed, conservationists
113 still have a very limited understanding of species and community responses to habitat change,
114 and our knowledge is largely derived from a small number of sites (Barlow et al. 2007; T.
115 Gardner et al. 2009, 2010). Furthermore, few researchers have investigated the impacts of
116 subsistence activities on biodiversity (Borghesio 2008; Brown et al. 2013).

117

118 Madagascar is an example of a biodiversity-rich tropical developing country that is
119 expanding its protected area system through the creation of new multiple-use sites. The island
120 is a global conservation priority, boasting an unparalleled combination of species diversity
121 and endemism (Brooks et al. 2006), with the majority of its endemic biota being forest
122 dependent (Goodman and Benstead 2005). However, less than 16 % of the country retained
123 forest cover by 2000 (Harper et al. 2007; McConnell and Kull 2014). Since 2003,
124 Madagascar has been in the process of tripling the coverage of its protected area system, from
125 1.7 to over 6 million ha, in response to lobbying from international conservation

126 organisations and funders (Corson 2014). Known as the ‘Durban Vision’ after the location of
127 the fifth World Parks Congress at which it was launched, this ambitious programme has
128 necessitated modifications to the country’s conception of protected areas and their
129 governance. Previously, all protected areas were governed by the State, managed by the para-
130 statal Madagascar National Parks, and comprised only strict categories (I, II and IV;
131 Randrianandianina et al. 2003). Most of the new protected areas established as part of the
132 Durban Vision are co-managed by non-governmental organisations (NGOs) and local
133 communities, and are proposed or designated as categories V and VI (AGRECO 2012;
134 Gardner 2011; Virah-Sawmy et al. 2014), with zoned areas where subsistence and low-level
135 commercial natural resource use activities are permitted (e.g. Gardner et al. 2008; Virah-
136 Sawmy et al. 2014; WWF 2010).

137

138 The goals of the expanded Madagascar Protected Area System (SAPM) are to conserve the
139 country’s unique biodiversity and its cultural heritage, as well as promoting the sustainable
140 use of natural resources for poverty alleviation and development (Commission SAPM 2006).
141 The simultaneous achievement of these goals is particularly complex because most forms of
142 traditional land and resource use in Madagascar have negative impacts on biodiversity
143 (Gardner 2009, 2011; Irwin et al. 2010). Planning the management of new multiple-use
144 protected areas requires an understanding of species and community responses to habitat
145 degradation arising from permitted resource use, yet our knowledge of the influence this has
146 on biodiversity is patchy for the country as a whole, and particularly for the globally-
147 important spiny forest ecoregion (Irwin et al. 2010). Moreover, existing studies in
148 Madagascar tend to mirror patterns in global research (Burivalova et al. 2014) by
149 summarising assemblage-level change via species richness (e.g. Randriamiharisoa et al.
150 2015; Scott et al. 2006). In other words, while studies may investigate the ecological or other

151 attributes of species remaining in degraded habitats, their results are usually reported in terms
152 of species richness, but this measure has been criticised because it can mask community
153 turnover from specialists to generalists (Barlow et al. 2007; Gardner et al. 2010). Here we
154 investigate bird and reptile community responses to habitat change in a new protected area in
155 the spiny forest ecoregion to ascertain the impacts of permitted and illegal livelihood
156 activities (charcoal production and shifting cultivation respectively) on the conservation value
157 of the vertebrate fauna. To overcome the issues associated with species richness as a metric,
158 we use a novel Conservation Value Index (CVI) to examine the influence of habitat
159 degradation on the two taxonomic assemblages.

160

161 **2 Methods**

162 **2.1 Study site**

163 Madagascar's spiny desert (or spiny forest), is a global priority ecoregion (Olson and
164 Dinerstein 1998) and Endemic Bird Area (Stattersfield et al. 1998) with extremely high rates
165 of local floral endemism (Phillipson 1996). Between 1990 and 2010 it suffered the fastest
166 rates of deforestation of any ecoregion in the country (Harper et al. 2007; ONE et al. 2013)
167 and, prior to 2003, it was the least represented ecoregion within the country's protected area
168 network (Fenn 2003).

169

170 Ranobe PK32 is a new protected area that received temporary protected status within the
171 Durban Vision framework in 2008, and is co-managed by local community associations, the
172 regional Forest Service and the international NGO WWF (Virah-Sawmy et al. 2014). Lying
173 north of the regional capital Toliara between the Fiherenana and Manombo rivers (Fig. 1), it
174 is the richest landscape in the ecoregion in terms of its bird, reptile and lemur fauna (Gardner
175 et al. 2009a,b; 2015a). However, the area is inhabited by approximately 90,000 people (WWF

176 2010), many of whom depend on natural resources from within and around the protected area
177 for their subsistence and household income (Gardner and Davies 2014; Gardner et al. 2015b).
178 Ranobe PK32 is thus proposed as a category VI protected area in which subsistence and low-
179 level commercial livelihood activities (such as timber cutting, fuelwood collection and
180 charcoal production, grazing and the harvesting of NTFPs) are permitted in sustainable use
181 zones which cover 86.5 % of the protected area's 148,554 ha (Virah-Sawmy et al. 2014;
182 WWF 2010).

183
184 Charcoal is primarily produced in the western part of the protected area, due to the presence
185 of the Route Nationale 9 (RN9) road that facilitates transportation. The industry is driven by
186 the close proximity of Toliara, a city of approximately 200,000 people in which 98 % of
187 households use wood or charcoal for cooking; demand from the city tripled between 2000
188 and 2007, and is largely met by anarchic charcoal production along the RN9 (Gardner et al.
189 2015b; Partage 2008). Since the region lacks fuelwood plantations, charcoal is produced
190 entirely from natural forests (Bertrand et al. 2010). Charcoal producers select only hardwood
191 trees (Randriamalala et al. 2016), thus causing forest degradation rather than outright
192 deforestation (Casse et al. 2004).

193
194 We conducted our study in the vicinity of Ranobe, a complex of three villages with a total
195 population of approximately 2000 people (Gardner and Davies 2014), where the surrounding
196 forests had been subjected to both charcoal production and shifting cultivation within recent
197 years. We selected three areas within 3 km of the main village which, until recently, were
198 part of a contiguous and relatively homogeneous forest block. Subsequently, the three areas
199 have suffered varying levels of disturbance that are indicative of the habitat degradation
200 gradient found across the whole landscape: i) a forest area showing minimal impacts of

201 human activity (low-disturbance, hereafter *Low*); ii) a forest area subject to intensive charcoal
202 production (moderate-disturbance, *Mod*); and, iii) an area regenerating following shifting
203 cultivation (high-disturbance, *High*). While *Low* and *Mod* retained a complex three-
204 dimensional structure and can be termed forest, *High* was an open area dominated by shrubs,
205 with only scattered trees (Fig. 1, Table 1). As there were no areas of forest near Ranobe that
206 had not been subject to any human disturbance, it was not possible to include a control site
207 representing intact habitat. Birds and reptiles were surveyed between January and March
208 2010 in the rainy season, when both groups are most active (Glaw and Vences 2007; Safford
209 and Hawkins 2013).

210

211 [Fig. 1]

212

213 [Table 1]

214

215 **2.2 Bird survey protocol**

216 We established 48 census stations within each area and used the point count method (Bibby
217 et al. 1998) to estimate bird relative abundance. Access to the forest interior was hindered by
218 the impenetrable nature of the vegetation at *Low* and *Mod*, so census stations were placed on
219 a stratified random grid along existing ox-cart tracks. We positioned all stations at a
220 perpendicular distance of 75 m from a track (following Jones et al. 1995) to minimise the
221 influence of edge effects, and at least 150 m apart to minimise the risk of double counting.

222

223 We surveyed each census station for 15 minutes (following a settling period of four minutes
224 after arrival), during which we recorded all visual and auditory contacts within 50 m of the
225 census station. To reduce time-of-day and weather-related effects, surveys were limited to

226 between 06.00 and 08.00 and were not conducted on rainy or windy days. The majority of
227 bird contacts in spiny forest (> 85 % at *Low* and *Mod*) were auditory due to the dense
228 vegetation, thus making it difficult to generate reliable distance estimates for bird contacts
229 and, as such, we did not employ distance sampling methods. However, the non-visual nature
230 of most contacts reduces the likelihood of a detectability bias arising from surveying in
231 forests of varying degradation levels (Bibby and Buckland 1987). The auditory nature of
232 most contacts also meant that we could not accurately count the number of individuals for
233 social species, and thus we recorded the presence of groups not individuals. We did not
234 include contacts with juvenile birds in our data analysis to reduce seasonality effects. Point
235 count observations yielded both relative frequency (defined as the proportion of counts in
236 which a given species was recorded) and relative abundance (mean number of contacts of a
237 given species per count) data.

238

239 **2.3 Reptile survey protocol**

240 We calculated the relative abundance of reptiles based on capture in pitfall traps and area
241 constrained refuge searches (transects), because observation and capture-based methods
242 permit the sampling of different components of the reptile fauna (Raselimanana 2008). For
243 pitfall trapping we followed a standard protocol widely used in Madagascar (D’Cruze et al.
244 2007; Raselimanana 2008). The traps consisted of plastic buckets (270 mm deep, 290 mm
245 internal diameter at top, 220 mm internal diameter at base) placed 10 m apart and buried in
246 the ground with the rim level with the surface. Drainage holes were drilled in the bottom of
247 each bucket and the handles were removed. Buckets were connected by a drift fence 500 mm
248 high, passing directly over the centre of each bucket, constructed from a sheet of plastic
249 supported by wooden stakes. The lower 50 mm of the fence was buried in the soil and
250 covered with leaf litter to prevent animals passing underneath. Within each treatment we

251 established three trap lines (each of 10 or 11 buckets), placed randomly, but at least 150 m
252 apart. Traps were constructed in the morning and left open for 13 nights, equating to 403 trap
253 nights in total per area, and were checked at 07.00 and 16.00 each day. All captured animals
254 were marked on the hind leg or ventral surface with nail polish, and released at the site of
255 capture. Recaptured individuals were excluded from the data analysis.

256

257 We also established 38 transects along which we conducted active refuge searches. Each
258 transect consisted of a 50 m rope erected adjacent to forest tracks based on a stratified
259 random grid. Each transect was at least 150 m apart, ran perpendicular to a track and started
260 10 m into the forest to reduce the influence of edge effects. We established each transect 24
261 hours prior to surveying to minimise disturbance effects. During surveys, two observers
262 moved slowly along each transect and searched for reptiles within 2 m of the central line,
263 scanning the trunks and branches of trees, searching within tree holes, under bark, in the leaf
264 litter and under/within dead branches. All reptiles initially observed within 2 m of the central
265 line were recorded. Transects were walked from 08.00-10.00 (n = 22/site) and 15.00-17.00 (n
266 = 16/site); we did not survey during periods of rain or thick cloud cover to minimise variation
267 in weather-related detectability, which reduced the number of appropriate afternoon survey
268 periods. Juveniles were omitted from the dataset to minimise any bias that might be
269 associated with the effects of the breeding season. Transects and pitfalls generated density
270 and capture rate data, respectively: we pooled the data and used total contacts for further
271 analyses (not including rarefaction).

272

273 **2.4 Data analysis**

274 In order to compare species richness between treatments and estimate the completeness of
275 our sampling, we generated individual-based observed species richness rarefaction curves

276 and associated 95 % confidence intervals using EstimateS v.9.0 (Colwell 2013). For reptiles,
277 we combined the two datasets by assigning species to one or other method on the basis of
278 substrate use, following a protocol adapted from Bicknell et al. (2015), whenever a species
279 was recorded by both methods. Thus all arboreal species were assigned to transects and all
280 terrestrial and fossorial lizards were assigned to pitfall traps. Remaining terrestrial species
281 (snakes and a tortoise) were assigned to the method by which they were most frequently
282 recorded. We used chi-squared contingency tables to test for homogeneity of observed
283 species relative frequency (birds) or total contacts (reptiles) across treatments.

284

285 *Conservation Value Index*

286 All species are not equal, and may differ in their value to conservationists on the basis of
287 endemism, extinction risk (Mace et al. 2008), evolutionary distinctiveness (Tucker et al.
288 2012; Hidasi-Neto et al. 2015), public appeal (Smith et al. 2012) or other attributes
289 (Humphries et al. 1995; Joseph et al. 2009). This variation forms the basis of a range of
290 protocols designed to elucidate the conservation value of species and, in turn, support the
291 prioritisation of conservation actions or funding allocations (e.g. Huang et al. 2016; Isaac et
292 al. 2007; Joseph et al. 2009). However, these protocols tend to be methodologically complex
293 and require the collection of large datasets, diminishing their applicability in situations where
294 non-academic conservation practitioners lack the training to apply them or where the
295 necessary data are unavailable (Gardner et al. 2015a). In this study, we therefore used the
296 novel Conservation Value Index (CVI) (adapted from Gardner et al. 2015a) to quantify the
297 conservation value of individual species because it uses only readily available data and does
298 not require the use of specialist software. As such, it can be easily applied in day-to-day
299 decision-making by conservation practitioners. We assigned CVI scores to individual species

300 of bird and reptile before combining them to assess the impacts of natural resource use, and
301 subsequent habitat degradation, on the conservation value of spiny forest habitats.

302

303 For the CVI we assigned scores to each individual species based on four attributes that reflect
304 rarity, distinctiveness and threat. We use different combinations of attributes for the two
305 taxonomic groups because the variation in conservation value within each group is driven by
306 different factors. We scored rarity using geographical scale of endemism (*G*) and
307 representation within SAPM (*R*), distinctiveness by taxonomic level of endemism (*E*), and
308 threat on the basis of hunting and collection pressure (*C*) and degradation tolerance (*T*). We
309 did not use *E* for reptiles because all species are endemic and there are no endemic families,
310 so variation in the attribute is limited. Similarly, we did not use *C* for birds because most
311 species in the Ranobe area are subject to comparable hunting pressure (Gardner and Davies
312 2014).

313

314 Introduced species were removed from the dataset and scores assigned to indigenous taxa on
315 a scale of 1-5 for each attribute (Table 2). For *G* we used different scoring systems for
316 reptiles and birds because species distributions of the two taxonomic groups are best
317 explained by different biogeographical models (Pearson and Raxworthy 2009). For birds we
318 used distribution maps from Safford and Hawkins (2013) and followed Stattersfield et al.
319 (1998) to classify microendemic species, whereas for reptiles we visually estimated range
320 criteria using maps in Glaw and Vences (2007) and adopted 10,000 km² as the threshold for
321 microendemic species (following Gardner et al. 2015a). *E* was assigned on the basis of
322 taxonomy in Safford and Hawkins (2013), *R* scores were assigned on the basis of occurrence
323 in 14 (birds) or 15 (reptiles) protected areas in the dry regions of Madagascar derived from
324 the literature (Online resource 1), and values for *C* were based on occurrence in CITES

325 (Convention on International Trade in Endangered Species) appendices and the literature on
326 reptile declines in Madagascar. *T* was attributed following the methods outlined in Gardner et
327 al. (2015a) for reptiles, and were based on the literature (Safford and Hawkins 2013; Wilmé
328 1996) for birds. Species for which no degradation tolerance data were available were scored
329 as intolerant on the basis of the precautionary principle.

330

331 [Table 2]

332

333 The individual species CVI scores were calculated, producing a value in the range of 4-100,
334 using the following formulae for reptiles and birds:

335
$$CVI_{reptile} = (G + R) \times (C + T)$$

336
$$CVI_{birds} = (G + E) \times (R + T)$$

337

338 The conservation value of a site can be considered a function of: i) the value of the species
339 occurring there; and, ii) their abundance, because an area with a large population of a
340 valuable species is more important than one with a small population. To understand the
341 relative conservation value of each habitat treatment, we therefore wanted a metric that
342 combined the CVI of each species with their relative abundance. However, simply weighting
343 the CVI score by the relative frequency would heavily bias common species at the expense of
344 rarer ones which are recorded only infrequently. We thus gave each species weightings
345 standardised to the treatment where it was most frequent (e.g., a species with relative
346 frequency of 0.36, 0.18 and 0.12 across each of the three treatments would be given
347 weightings of 1, 0.5 and 0.33 respectively). In each treatment the CVI was then multiplied by
348 the weighting to produce a frequency-weighted CVI score for each species, before these were
349 summed to produce a conservation value score for each treatment.

350

351 **3 Results**

352 **3.1 Degradation impacts on birds**

353 We recorded 2385 bird contacts, comprising 53 species, in point counts across all treatments.
354 Rarefaction curves approach an asymptote in all treatments, indicating that bird communities
355 were sufficiently sampled (Fig. S1). Although observed richness was highest in the moderate-
356 degradation treatment (*Low* – 36 spp.; *Mod* – 43 spp.; *High* – 37 spp.), rarefaction curves
357 show no significant differences in richness since the 95 % confidence intervals overlap
358 (Online resource 2). Total richness is estimated at 42.0 (*Low*), 46.8 (*Mod*) and 39.7 (*High*)
359 species in the three treatments. Twenty-four species (45.3 %) were recorded in all treatments,
360 one species (1.9 %) was restricted to *Low*, five species (9.4 %) were restricted to *Mod*, and
361 seven (13.2 %) species were restricted to *High*: 17 species (32.1 %) were recorded only in
362 forest habitats (*Low* and *Mod*).

363

364 Observed patterns of species relative frequency differed significantly for 22 species (41.5 %)
365 across the three treatments. Three of these species (*Cuculus rochii*, *Hypsipetes*
366 *madagascariensis* and *Dicrurus forficatus*) were observed more frequently in the low-
367 degradation treatment, one species (*Ploceus sakalava*) in the moderate-degradation treatment,
368 and six species (*Turnix nigricollis*, *Oena capensis*, *Agapornis canus*, *Cisticola cherina*,
369 *Acridotheres tristis* and *Foudia madagascariensis*) in the high-degradation treatment. A
370 further 12 species were recorded less frequently in the high-degradation treatment than in
371 forest habitat (*Low* or *Mod*) (Online resource 3).

372

373 Patterns of species endemism varied across the degradation gradient (Fig. 2). While the
374 proportion of endemic species was approximately equal in all treatments, the high-

375 degradation treatment contained a lower proportion of regionally-endemic birds (defined as
376 restricted to Madagascar and the islands of the western Indian Ocean) and a higher proportion
377 of non-endemic species. The vast majority (97.9 %) of contacts with introduced species
378 (*Acridotheres tristis*) occurred in the high-degradation treatment.

379

380 [Fig. 2]

381

382 **3.2 Degradation impacts on reptiles**

383 We recorded 661 reptile contacts comprising 32 species, 27 of which were recorded at *Low*
384 and *Mod*, and 15 species at *High*. Twenty-two species were observed during transects, and 27
385 were captured in pitfall traps (Online resource 4). Twelve species (37.5 %) were recorded in
386 all treatments, 17 species (53.1 %) were only recorded in forest habitats, and one species
387 (*Lygodactylus tuberosus*) was recorded only in the high-disturbance site. Rarefaction curves
388 indicate that *Low* and *Mod* had significantly higher species richness than *High*, as there is no
389 overlap between confidence intervals (Online resource 5). Total richness is estimated at 30.5
390 (*Low*), 34.2 (*Mod*) and 19.1 (*High*) species in the three treatments.

391

392 Observed patterns of reptile abundance, based on total contacts, were significantly
393 heterogeneous for 11 species (34.4 %). Three species were recorded more frequently in the
394 low-degradation treatment (*Chalarodon madagascariensis*, *Lygodactylus verticillatus* and
395 *Oplurus cyclurus*), two species in the moderate-degradation treatment (*Madascincus* cf.
396 *igneocaudatus* and *Tracheloptychus petersi*), and three species in the high-degradation
397 treatment (*Lygodactylus tuberosus*, *Paroedura picta* and *Typhlops arenarius*). A further three
398 species (*Geckolepis* c.f. *polypelis*, *Phelsuma mutabilis* and *Trachylepis elegans*) were
399 recorded more frequently in the two forest areas than in the high-degradation treatment.

400

401 Forest disturbance affected distinct components of the reptile community differently,
402 depending on their foraging substrate (Online resource 6). Terrestrial species decreased in
403 frequency (capture rate and/or density) with increasing disturbance, while arboreal species
404 demonstrated reduced frequency at *Mod* and reduced richness at *High* compared to the less
405 degraded site. Fossorial and litter dwelling species reached peak frequency under conditions
406 of moderate-intensity disturbance.

407

408 **3.3 Conservation value of species and sites**

409 The CVI allowed us to weight species on the basis of their conservation value. The six
410 highest scoring bird species were locally-endemic forest specialists (Table 3), while the
411 highest scoring reptile was the heavily harvested (and thus Critically Endangered) tortoise
412 *Pyxis arachnoides* (Table 4). The relative conservation value of each treatment varied for the
413 two taxonomic groups. Total bird conservation value was highest in *Low*, while total reptile
414 conservation value was highest in *Mod*, although in both cases the differences between the
415 two forest areas were small (Table 5). The high-degradation treatment had the lowest
416 conservation value for both taxa.

417

418 [Table 3]

419

420 [Table 4]

421

422 [Table 5]

423

424 **4 Discussion**

425 We have generated some of the first data on the impacts of permitted livelihood activities
426 within Madagascar's new generation of multiple-use protected areas. Our results show that
427 charcoal production, an authorised activity within much of the Ranobe PK32 protected area,
428 resulted in an overall reduction in the conservation value of habitats, although the responses
429 of reptile and bird communities varied. However the impacts of charcoal production were less
430 severe than the impacts of illegal shifting cultivation for both groups.

431

432 Although the impacts of habitat degradation on Madagascar's biodiversity have been well
433 studied (reviewed in Gardner 2009; Irwin et al. 2010), the vast majority of research has been
434 conducted in the country's humid and dry forests, ecosystems which greatly differ from the
435 spiny forest in terms of biotic communities and abiotic conditions (Moat and Smith 2007;
436 Goodman and Raherilalao 2013). Within the spiny forest, degradation has been found to
437 reduce species richness in both birds (Randriamiharisoa et al. 2015) and reptiles (Theisinger
438 and Ratianarivo 2015). However, in our study, richness was maintained for reptiles and
439 increased for birds at moderate degradation levels. Indeed the conservation value of reptiles
440 was greatest at the moderate-disturbance site, perhaps reflecting an increase in microhabitat
441 heterogeneity or structural complexity (MacArthur and MacArthur 1961; Tews et al. 2004).
442 Bird communities were more responsive than reptiles to habitat degradation, undergoing
443 extensive community turnover. This was reflected in the greater prevalence of birds adapted
444 to open areas, and a decrease in the frequency of certain high-value, locally-endemic species
445 such as *Monias benschi*, *Coua cursor* and *Newtonia archboldi*, with increasing degradation
446 intensity.

447

448 Wilmé (1996) suggests that "the tolerance of [Madagascar's] endemic forest avifauna to
449 forest degradation is proportional to its degree of taxonomic endemism". However, we

450 recorded seven members of endemic genera (*Coua cursor*, *Monias benschi*, *Neomixis*
451 *striatigula*, *Newtonia brunneicauda*, *N. archboldi*, *Vanga curvirostris* and *Xenopirostris*
452 *xenopirostris*) previously thought to occur only in undisturbed or slightly disturbed habitats,
453 within a largely deforested habitat in our high-disturbance treatment. These findings lend
454 some support to the hypothesis that faunal species of Madagascar's dry and spiny forests may
455 be more tolerant of degradation than those same or congeneric species in the country's humid
456 east and north (Gardner 2009). This may arise due to the more 'gentle' habitat modifications
457 occurring in dry forests compared to rainforests (Irwin et al. 2010): for example, the
458 increased light penetration in forest gaps is thought to make little difference to the understory
459 in the spiny forest, because the sparse, deciduous nature of the canopy already allows
460 illumination at ground level (Seddon and Tobias 2007). However, while tropical dry forests
461 are thought to be more resilient than humid forests in terms of regeneration capacity (Lebrija-
462 Trejos et al. 2008), little is known about the relative disturbance sensitivity of their respective
463 faunas. Such research should be considered a priority since it has important repercussions for
464 the implementation of multiple-use protected areas in different bioclimatic contexts.

465

466 The finding that moderate levels of degradation provoked an increase in richness of birds,
467 and maintained richness in reptiles, is consistent with Connell's (1978) 'intermediate
468 disturbance hypothesis', and reflects a pattern widely reported from other tropical
469 environments, at least for some guilds (Burivalova et al. 2014; Child et al. 2009; Gray et al.
470 2007; Martin and Blackburn 2010; Pons and Wendenberg 2005). However, all species are not
471 equal, and the greater richness may often mask a turnover from range-restricted specialists to
472 widespread generalists (Canaday 1997; Christian et al. 2009; Holbech 2005; Petit and Petit
473 2003; Scott et al. 2006). The latter are of less importance to conservationists precisely
474 because they adapt well to anthropogenic disturbance and thus do not require conservation

475 actions, such as protected areas, to maintain them (Harris and Pimm 2004; T. Gardner et al.
476 2009). The use of species richness as a measure of conservation value has been widely
477 criticised for this reason (Barlow et al. 2007; DeClercke et al. 2010; Fermon et al. 2005;
478 Norris et al. 2010), but remains persistent (e.g., studies reviewed by Burivalova et al. 2014).
479 Our use of the CVI provides further evidence of the inadequacies of richness in prioritising
480 between sites or habitats, as the use of richness would indicate that forests degraded by
481 charcoal production are more valuable for bird conservation in the spiny forest than less
482 degraded habitats. Of course, the CVI does not represent a definitive quantification of
483 conservation value, but is a useful heuristic tool to help conservationists prioritise action to
484 where it is most needed (i.e. high-value species), and can be used without training, complex
485 software or collecting new data.

486

487 Although the use of CVI provides novel insights into the impacts of habitat change on the
488 conservation value of spiny forest bird and reptile assemblages, our results must be
489 interpreted with caution. We carried out surveying during the rainy season when both groups
490 are most active, and surveyed each site sequentially for logistical reasons. However, biases
491 may have arisen due to changes in species detectability related to the advancing breeding
492 season. In addition, the entry of new cohorts may have increased population size as surveying
493 progressed. We minimised the latter problem by excluding all records of juveniles from the
494 analysis, although it would have been preferable to repeat data collection over multiple years,
495 or to survey each site simultaneously using multiple teams. Nonetheless, the latter approach
496 has a number of drawbacks, including the extensive training needed to minimise the biases
497 associated with potential differences in the bird detection abilities and/or identification skills
498 of research assistants.

499

500 Although our observations appear to suggest that the majority of bird and reptile species in
501 Ranobe are somewhat resilient to moderate or high levels of degradation, the presence of a
502 species does not necessarily equate to its viability. It should not be assumed that local
503 populations in disturbed areas will persist in the long-term because there are likely to be time
504 lags associated with the impacts arising from perturbation, meaning that the degraded habitats
505 at Ranobe may be carrying an ‘extinction debt’ (Kuussaari et al. 2009; Tilman et al. 1994).
506 This is particularly true given that the habitat modifications that are the focus of this study are
507 relatively recent (range: 3-15 years across the treatments). In addition, the persistence of
508 some species within degraded habitats may be the result of source-sink dynamics, with
509 populations maintained only by immigration from nearby areas of higher quality habitat
510 (Hylander and Ehrlén 2013; Pulliam 1988; Tilman et al. 1994). The degraded habitats at
511 Ranobe may therefore experience future local extinctions, even without further modification,
512 and we may have over-estimated the value of these areas for bird and reptile diversity
513 (Barlow et al. 2007; Sekercioglu et al. 2007). The scale of extinction debt can be influenced
514 by habitat quantity, quality, or connectivity (Hylander and Ehrlén 2013). As such, when
515 destructive activities such as charcoal production cannot be prevented within the ‘sustainable
516 use zones’ of multiple-use protected areas, both the structural and functional connectivity
517 between high-quality habitat patches should be maximised in order to maintain biodiversity
518 and mitigate the negative impacts associated with resource exploitation.

519

520 The suggestion that Madagascar’s new generation of multiple-use protected areas may suffer
521 the continued erosion of biodiversity as a result of the impacts of authorised livelihood
522 activities has important ramifications for the objectives and management of multiple-use sites
523 worldwide. In a multi-taxon assessment across a continuum of protection levels in East
524 Africa, Gardner et al. (2007) found that multiple-use protected areas provide significant and

525 complementary conservation services to strictly-protected sites, maintaining species richness
526 but conserving significantly different faunal communities to those occurring in national
527 parks. Thus a spectrum of protected area categories may be appropriate to conserve the full
528 complement of biodiversity in continental regions, if these possess a range of faunal
529 assemblages adapted to a continuum of habitat types from dense forests to wooded savannahs
530 and grasslands (Borghesio 2008; Gardner et al. 2007; Pons et al. 2003).

531

532 Madagascar, however, differs from continents in that the vast majority of the endemic biota is
533 forest-dependent (Goodman and Benstead 2005), while non-forest areas typically contain
534 floristically- and faunistically-impoverished assemblages characterised by non-endemic
535 species of low conservation value (Irwin et al. 2010; Koechlin et al. 1974; Lowry II et al.
536 1997). In this context, multiple-use sites essentially conserve the same communities as strict
537 protected areas, but may do so less successfully than the latter. Thus, while multiple-use
538 categories may be the only politically, ethically and logistically feasible option for many of
539 Madagascar's new generation of protected areas, given the socioeconomic importance to
540 rural communities of remaining forest resources (Gardner et al. 2013), it should not be
541 assumed that they will be successful in maintaining the biodiversity they were established to
542 conserve. Given that range-restricted habitat specialists are disproportionately likely to go
543 extinct in modified habitats (Posa and Sodhi 2006; Scales and Marsden 2008), and are of
544 greatest conservation interest worldwide, careful attention must be paid to the choice of
545 protected area models in different contexts; in regions where the majority of priority species
546 are disturbance-sensitive, strict protected areas may be a more appropriate model if they can
547 be managed effectively.

548

549 **Acknowledgements**

550 This research was funded by an African Bird Club conservation grant to CJG. We would like
551 to thank the community of Ranobe for hosting us during our research. In addition, we are
552 grateful to E. Rasolondranaly, B. Rasolonandrasana and M. Virah-Sawmy for logistical
553 support, and B. Morgan and J.E. Bicknell for statistical advice.

554

555 **References**

556 Adams WM, Hutton J (2007) People, parks and poverty: political ecology and biodiversity
557 conservation. *Conserv Soc* 5: 147–183.

558

559 AGRECO (2012) Analyse des coûts et sources de financement du système des aires
560 protégées de Madagascar (Octobre 2010 – Janvier 2012). AGRECO, Antananarivo.

561

562 Barlow J, Gardner TA, Araujo IS, Avila-Pires TC, Bonaldo AB, Costa JE, Esposito MC,
563 Ferreira LV, Hawes J, Hernandez MIM et al (2007) Quantifying the biodiversity value of
564 tropical primary, secondary, and plantation forests. *PNAS* 104: 18555–18560.

565

566 Barnosky AD, Matzke N, Tomiya S, Wogan GOU, Swartz B, Quental TB, Marshall C,
567 McGuire JL, Lindsey EL, Maguire KC, et al. (2011) Has the Earth’s sixth mass extinction
568 already arrived? *Nature* 471: 51-57

569

570 Beaudrot L, Ahumada JA, O’Brien T, Alvarez-Loayza P, Boekee K, Campos-Arceiz A,
571 Eichberg D, Espinosa S, Fegraus E, Fletcher C, et al. (2016) Standardized assessment of
572 biodiversity trends in tropical forest protected areas : the end is not in sight. *PLoS Biol* 14:
573 e1002357.

574

575 Bertrand A, Ramamonjisoa B, Montagne P (2010) Les filières péri-urbaines
576 d’approvisionnement en bois énergie des grandes villes de Madagascar. In Montagne P,
577 Razafimahatratra S, Rasamindisa A, Crehay R (eds) ARINA, le charbon de bois à
578 Madagascar: entre demande urbaine et gestion durable. CITE, Antananarivo, pp 23–36.

579

580 Bertzky B, Corrigan C, Kemsey J, Kenney S, Ravilious C, Besançon C, Burgess N (2012)
581 Protected planet report 2012: tracking progress towards global targets for protected areas.
582 IUCN, Gland and UNEP-WCMC, Cambridge.

583

584 Bibby CJ, Buckland ST (1987) Bias of bird census results due to detectability varying with
585 habitat. *Acta Oecol* 8: 103–112.

586

587 Bibby C, Jones M, Marsden S (1998) Expedition field techniques: bird surveys. Royal
588 Geographical Society, London.

589

590 Bicknell JE, Struebig MJ, Davies ZG (2015) Reconciling timber extraction with biodiversity
591 conservation in tropical forests using reduced-impact logging. *J Appl Ecol* 52: 379–388.

592

593 Borghesio L (2008) Effects of human subsistence activities on forest birds in northern Kenya.
594 *Conserv Biol* 22: 384–394.

595

596 Brooks TM, Mittermeier RA, da Fonseca GAB, Gerlach J, Hoffmann M, Lamoreux JF,
597 Mittermeier CG, Pilgrim JD, Rodrigues ASL (2006) Global biodiversity conservation
598 priorities. *Science* 313: 58–61.

599

600 Brown KA, Johnson SE, Parks KE, Holmes SM, Ivoandry T, Abram NK, Delmore KE,
601 Ludovic R, Andriamaharoa HE, Wyman TM et al. (2013) Use of provisioning ecosystem
602 services drives loss of functional traits across land use intensification gradients in tropical
603 forests in Madagascar. *Biol Conserv* 161: 118–127.

604

605 Burivalova Z, Şekercioğlu CH, Koh LP (2014) Thresholds of logging intensity to maintain
606 tropical forest biodiversity. *Curr Biol* 24: 1893–1898.

607

608 Butchart SHM, Clarke M, Smith RJ, Sykes RE, Scharlemann JPW, Harfoot M, Buchanan
609 GM, Angulo A, Balmford A, Bertzky B, et al. (2015) Shortfalls and solutions for meeting
610 national and global conservation area targets. *Conserv Lett* 8: 329–337.

611

612 Cabeza M (2013) Knowledge gaps in protected area effectiveness. *Anim Conserv* 16: 381–
613 382.

614

615 Canaday C (1997) Loss of insectivorous birds along a gradient of human impact in
616 Amazonia. *Biol Conserv* 77: 63–77.

617

618 Casse T, Milhøj A, Ranaivoson S, Randriamanarivo JR (2004) Causes of deforestation in
619 southwestern Madagascar: what do we know? *For Policy Econ* 6: 33–48.

620

621 CBD (Convention on Biological Diversity) (2010) Decision adopted by the Conference of the
622 Parties to the Convention on Biological Diversity at its tenth meeting [Decision X/2] Nagoya,

623 Aichi Prefecture, Japan, 18–29 October 2010. Secretariat to the Convention on Biological
624 Diversity, Montreal.

625

626 Child MF, Cumming GS, Amano T (2009) Assessing the broad-scale impact of agriculturally
627 transformed and protected area landscapes on avian taxonomic and functional richness. *Biol*
628 *Conserv* 142: 2593–2601.

629

630 Christian K, Isabelle LV, Frédéric J, Vincent D (2009) More species, fewer specialists: 100
631 years of changes in community composition in an island biogeographical study. *Diversity and*
632 *Distributions* 15: 641–648.

633

634 Colwell RK (2013) EstimateS: Statistical estimation of species richness and shared species
635 from samples. Version 9.0.

636

637 Commission SAPM (2006) *Système d’aires protégées Malagasy: orientations générales sur*
638 *les catégories et les types de gouvernance*. SAPM Commission, Antananarivo.

639

640 Connell JH (1978) Diversity in tropical rainforests and coral reefs. *Science* 199: 1302–1310.

641

642 Cornet A (1974) *Essai de Cartographie Bioclimatique à Madagascar*. Orstom, Paris.

643

644 Corson C (2014) Conservation politics in Madagascar: the expansion of protected areas. In
645 Scales I (ed) *Conservation and environmental management in Madagascar*. Routledge,
646 London, pp 193–215.

647

648 D’Cruze N, Sable J, Green K, Dawson J, Gardner C, Robinson J, Starkie G, Vences M, Glaw
649 F (2007) The first comprehensive survey of amphibians and reptiles at Montagne des
650 Français, Madagascar. *Herpetol Conserv Biol* 2: 87–99.
651

652 DeClerck FAJ, Chazdon R, Holl KD, Milder JC, Finegan B, Martinez-Salinas A, Imbach P,
653 Canet L, Ramos Z (2010) Biodiversity conservation in human-modified landscapes of
654 Mesoamerica: past, present and future. *Biol Conserv* 143: 2301–2313.
655

656 Dudley N (ed) (2008) *Guidelines for applying protected area management categories*. IUCN,
657 Gland, Switzerland.
658

659 Dudley N, Parrish JD, Redford KH, Stolton S (2010) The revised IUCN protected area
660 management categories: the debate and ways forward. *Oryx* 44: 485–490.
661

662 Dudley N, Groves C, Redford KH, Stolton S (2014) Where now for protected areas? Setting
663 the stage for the 2014 World Parks Congress. *Oryx* 48: 496–503.
664

665 Ehrlich PR, Pringle RM (2008) Where does biodiversity go from here? A grim business-as-
666 usual forecast and a hopeful portfolio of partial solutions. *PNAS* 105: 11579–11586.
667

668 Fenn MD (2003) The spiny forest ecoregion. In Goodman SM, Benstead JP (eds) *The natural*
669 *history of Madagascar*. University of Chicago Press, Chicago, pp 1525–1529.
670

671 Fermon H, Waltert M, Vane-Wright RI, Muhlenberg M (2005) Forest use and vertical
672 stratification in fruit-feeding butterflies of Sulawesi, Indonesia: impacts for conservation.
673 *Biodivers Conserv* 14: 333–350.

674

675 Ferraro PJ, Hanauer MH, Miteva DA, Canavire-Bacarreza GJ, Pattanayak SK, Sims KR
676 (2013) More strictly protected areas are not necessarily more protective: evidence from
677 Bolivia, Costa Rica, Indonesia and Thailand. *Environ Res Lett* 8: 025011.

678

679 Gardner CJ (2009) A review of the impacts of anthropogenic habitat change on terrestrial
680 biodiversity in Madagascar: Implications for the design and management of new protected
681 areas. *Malagasy Nature* 2: 2–29.

682

683 Gardner CJ (2011) IUCN management categories fail to represent new, multiple-use
684 protected areas in Madagascar. *Oryx* 45: 336–346.

685

686 Gardner CJ, Davies ZG (2014) Rural bushmeat consumption within multiple-use protected
687 areas: qualitative evidence from southwest Madagascar. *Hum Ecol* 42: 21–34.

688

689 Gardner CJ, Ferguson B, Rebara F, Ratsifandrihamanana AN (2008) Integrating traditional
690 values and management regimes into Madagascar's expanded protected area system: the case
691 of Ankodida. In Mallarach JM (ed) *Protected landscapes and cultural and spiritual values*.
692 Kasperek Verlag, Heidelberg, pp 92–103.

693

694 Gardner CJ, Kidney D, Thomas H (2009a) First comprehensive avifaunal survey of PK32-
695 Ranobe, a new protected area in south-western Madagascar. *Phelsuma* 17: 20–39.

696

697 Gardner CJ, Fanning E, Thomas H, Kidney D (2009b) The lemur diversity of the Fiherenana-
698 Manombo Complex, southwest Madagascar. *Madag Conserv Dev* 4: 38–43.

699

700 Gardner CJ, Nicoll ME, Mbohoahy T, Oleson KLL, Ratsifandrihamanana AN, Ratsirarson J,
701 René de Roland LA, Virah-Sawmy M, Zafindrasilivonona B, Davies ZG (2013) Protected
702 areas for conservation and poverty alleviation: experiences from Madagascar. *J Appl Ecol* 50:
703 1289–1294.

704

705 Gardner CJ, Raxworthy CJ, Metcalfe K, Raselimanana AP, Smith RJ Davies ZG (2015a)
706 Comparing methods for prioritising between existing protected areas: a case study using
707 Madagascar's dry forest reptiles. *PLoS ONE* 10: e0132803

708

709 Gardner CJ, Gabriel FUL, St John FAV, Davies ZG (2015b) Changing livelihoods and
710 protected area management: a case study of charcoal production in south-west Madagascar.
711 *Oryx* doi:10.1017/S0030605315000071

712

713 Gardner TA, Caro T, Fitzherbert EB, Banda T, Lalbhai P (2007) Conservation value of
714 multiple-use areas in East Africa. *Conserv Biol* 21: 1516–1525.

715

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

718

719 Gardner TA, Barlow J, Sodhi NS, Peres CA (2010) A multi-region assessment of tropical
720 forest biodiversity in a human-modified world. *Biol Conserv* 143: 2293–2300.

721

722 Gaston KJ, Jackson SF, Cantu-Salazar L, Cruz-Pinon G (2008) The ecological performance
723 of protected areas. *Annu Rev Ecol Evol Syst* 39: 93–113.

724

725 Geldmann J, Barnes M, Coad L, Craigie ID, Hockings M, Burgess N (2013) Effectiveness of
726 terrestrial protected areas in reducing habitat loss and population declines. *Biol Conserv* 161:
727 230–238.

728

729 Glaw F, Vences M (2007) A field guide to the amphibians and reptiles of madagascar, 3rd
730 edn. Vences and Glaw Verlags GbR, Köln.

731

732 Goodman SM, Benstead JP (2005) Updated estimates of biotic diversity and endemism for
733 Madagascar. *Oryx* 39: 73–77.

734

735 Goodman SM, Raherilalao MJ (2013) Atlas of selected land vertebrates of Madagascar.
736 Association Vahatra, Antananarivo.

737

738 Gray MA, Baldauf SL, Mayhew PJ, Hill JK (2007) The response of avian feeding guilds to
739 tropical forest disturbance. *Conserv Biol* 21: 133–141.

740

741 Harper GJ, Steininger MK, Tucker CJ, Juhn D, Hawkins F (2007) Fifty years of deforestation
742 and forest fragmentation in Madagascar. *Environ Conserv* 34: 325–333.

743

744 Harris GM, Pimm SL (2004) Bird species' tolerance of secondary forest habitats and its
745 effects on extinction. *Conserv Biol* 18: 1607–1616.

746
747
748
749
750
751
752
753
754
755
756
757
758
759
760
761
762
763
764
765
766
767
768
769
770

Hidasi-Neto J, Loyola R, Cianciaruso MV (2015) Global and local evolutionary and ecological distinctiveness of terrestrial mammals: identifying priorities across scales. *Diversity Distrib* 21: 548–559.

Holbech LH (2005) The implications of selective logging and forest fragmentation for the conservation of avian diversity in evergreen forests of south-west Ghana. *Bird Conserv Int* 15: 27–52.

Huang J, Lu X, Huang J, Ma K (2016) Conservation priority of endemic Chinese flora at family and genus levels. *Biodivers Conserv* 25: 23–35.

Humphries CJ, Williams PH, Vane-Wright RI (1995) Measuring biodiversity value for conservation. *Annu Rev Ecol Evol Syst* 26: 93–111.

Hylander K Ehlén J (2013) The mechanisms causing extinction debts. *Trends Ecol Evol* 28: 341–346.

Irwin MT, Wright PC, Birkinshaw C, Fisher B, Gardner CJ, Glos J, Goodman SM, Loiselle P, Rabeson P, Raharison JL et al (2010) Patterns of species change in anthropogenically disturbed habitats of Madagascar. *Biol Conserv* 142: 2351–2362.

Isaac NJB, Turvey ST, Collen B, Waterman C, Baillie JEM (2007) Mammals on the EDGE: conservation priorities based on threat and phylogeny. *PLoS ONE* 2: e296.

771 Jenkins CN, Joppa L (2009) Expansion of the global terrestrial protected area system. *Biol*
772 *Conserv* 142: 2166–2174.
773

774 Jones MJ, Linsley MD, Marsden SJ (1995) Population sizes, status and habitat associations of
775 the restricted-range bird species of Sumba, Indonesia. *Bird Conserv Int* 5: 21–52.
776

777 Joppa LN, Pfaff A (2011) Global protected area impacts. *Proc R Soc B* 278: 1633–1638.
778

779 Joseph LN, Maloney RF, Possingham HP (2009) Optimal allocation of resources among
780 threatened species: a project prioritisation protocol. *Conserve Biol* 23: 328–338.
781

782 Juffe-Bignoli D, Burgess ND, Bingham H, Belle EMS, de Lima MG, Deguidnet M, Bertzky
783 B, Milam AN, Martinez-Lopez J, Lewis E et al (2014) Protected Planet Report 2014.
784 Cambridge, UK: UNEP-WCMC.
785

786 Koechlin J, Guillaumet JL, Morat P (1974) Flore et végétation de Madagascar. J. Cramer
787 Verlag, Vaduz.
788

789 Kuussaari M, Bommarco R, Heikkinen RK, Helm A, Krauss J, Lindborg R, Öckinger E,
790 Pärtel M, Pino J, Rodà F et al (2009) Extinction debt: a challenge for biodiversity
791 conservation. *Trends Ecol Evol* 24: 564–571.
792

793 Lebrija-Trejos E, Bongers F, Pérez-García EA, Meave JA (2008) Successional change and
794 resilience of a very dry tropical deciduous forest following shifting agriculture. *Biotropica*
795 40: 422–431.

796

797 Leroux SJ, Krawchuk MA, Schmiegelow F, Cumming SG, Lisgo K, Anderson LG, Petkova
798 M (2010) Global protected areas and IUCN designations: do the categories match the
799 conditions? *Biol Conserv* 143: 609–616.

800

801 Locke H, Dearden P (2005) Rethinking protected area categories and the new paradigm.
802 *Environ Conserv* 32: 1–10.

803

804 Lowry PP, Schatz GE, Phillipson PB (1997) The classification of natural and anthropogenic
805 vegetation in Madagascar. In Goodman SM, Patterson BD (eds) *Natural change and human
806 impact in Madagascar*. Smithsonian Institution Press, Washington and London, pp 93–123.

807

808 MacArthur RH, MacArthur JW (1961) On bird diversity. *Ecology* 42: 594–598.

809

810 Mace GM, Collar NJ, Gaston KJ, Hilton-Taylor C, Akçakaya HR, Leader-Williams N,
811 Milne-Gulland EJ, Stuart SN (2008) Quantification of extinction risk: IUCN's system for
812 classifying threatened species. *Conserv Biol* 22: 1424–1442.

813

814 Martin TE, Blackburn GA (2010) Impacts of tropical forest disturbance upon avifauna on a
815 small island with high endemism: implications for conservation. *Conserv Soc* 8: 127–139.

816

817 Mascia MB, Paillet S, Krithivasan R, Roshchanka V, Burns D, Mlotha MJ, Murray DR, Peng
818 N (2014) Protected area downgrading, downsizing and degazettement (PADDD) in Africa,
819 Asia and Latin America and the Caribbean, 1900–2010. *Biol Conserv* 169: 355–361.

820

821 McConnell WJ, Kull CA (2014) Deforestation in Madagascar: debates over the island's
822 forest cover and challenges of measuring forest change. In Scales IR (ed) Conservation and
823 environmental management in Madagascar. Routledge, Abingdon, pp 65–104.
824

825 McDonald RI, Boucher TM (2011) Global development and the future of the protected area
826 strategy. *Biol Conserv* 144: 383–392.
827

828 Miller DC (2014) Explaining global patterns of international aid for linked biodiversity
829 conservation and development. *World Dev* 59: 341–359.
830

831 Moat J, Smith P (2007) Atlas of the vegetation of Madagascar. Royal Botanical Gardens,
832 Kew.
833

834 Montesino Pouzols F, Toivonen T, Di Minin E, Kukkala A, Kullberg P, Kuusterä J,
835 Lehtomäki J, Tenkanen H, Verburg PH, Moilanen A (2014) Global protected area expansion
836 is compromised by projected land-use and parochialism. *Nature* 516: 383–386.
837

838 Nelson A, Chomitz KM (2011) Effectiveness of strict vs. multiple use protected areas in
839 reducing tropical forest fires: a global analysis using matching methods. *PLoS ONE* 6:
840 e22722.
841

842 Norris K, Asase A, Collen B, Gockowski J, Mason J, Phalan B, Wade A (2010) Biodiversity
843 in a forest-agriculture mosaic – the changing face of West African rainforests. *Biol Conserv*
844 143: 2341–2350.
845

846 Olson DM, Dinerstein E (1998) The Global 200: a representation approach to conserving the
847 earth's most biologically valuable ecoregions. *Conserv Biol* 12: 502–515.
848

849 ONE (Office National pour l'Environnement), DGF/MEF, Conservation International, FTM,
850 Madagascar National Parks (2013) Evolution de la couverture de forêts naturelles à
851 Madagascar 2005-2010. Office National pour l'Environnement, Antananarivo.
852

853 Partage (2008). Analyse de l'offre et de la demande en énergie domestique au niveau de la
854 commune de Toliara. Partage, Antananarivo.
855

856 Pearson RG, Raxworthy CJ (2009) The evolution of local endemism in Madagascar:
857 watershed versus climatic gradient hypotheses evaluated by null biogeographic models.
858 *Evolution* 63: 959–967.
859

860 Peres CA, Barlow J, Laurance WF (2006) Detecting anthropogenic disturbance in tropical
861 forests. *Trends Ecol Evol* 21: 227–229.
862

863 Petit LJ, Petit DR (2003) Evaluating the importance of human-modified lands for neotropical
864 bird conservation. *Conserv Biol* 17: 687–694.
865

866 Phillipson PB (1996) Endemism and non-endemism in the flora of south-west Madagascar. In
867 Lourenço WR (ed) *Biogéographie de Madagascar*. Editions ORSTOM, Paris, pp 125–136.
868

869 Polak T, Watson JEM, Fuller RA, Joseph LN, Martin TG, Possingham HP, Venter O,
870 Cawardine J. (2015) Efficient expansion of global protected areas requires simultaneous
871 planning for species and ecosystems. *R Soc Open Sci* 2: 150107.
872

873 Pons P, Wendenbeurg C (2005) The impact of fire and forest conversion into savanna on the
874 bird communities of west Madagascan dry forests. *Anim Conserv* 8: 183–193.
875

876 Pons P, Lambert B, Rigolot E, Prodon R (2003) The effects of grassland management using
877 fire on habitat occupancy and conservation of birds in a mosaic landscape. *Biodivers Conserv*
878 12: 1843–1860.
879

880 Posa MRC, Sodhi NS (2006) Effects of anthropogenic land use on forest birds and butterflies
881 in Subic Bay, Philippines. *Biol Conserv* 129: 256–270.
882

883 Pulliam HR (1988) Sources, sinks, and population regulation. *Amer Nat* 132: 652–661.
884

885 Randriamalala JR, Radosy HO, Razanaka S, Randriambanona H, Hervé D (2016) Effects of
886 goat grazing and woody charcoal production on xerophytic thickets of southwestern
887 Madagascar. *J Arid Environ* 128: 65–72.
888

889 Randriamiharisoa LO, Rakotondravony D, Raherilalao MJ, Ranirison A, Wilmé L, Ganzhorn
890 JU (2015) Effects of transhumance route on the richness and composition of bird
891 communities in Tsimanampesotse National Park. *Madag Conserv Dev* 10: 110–115.
892

893 Randrianandianina BN, Andriamahaly LR, Harisoa FM, Nicoll ME (2003) The role of
894 protected areas in the management of the island's biodiversity. In Goodman SM, Benstead JP
895 (eds) *The natural history of Madagascar*. The University of Chicago Press, Chicago, pp
896 1423–1432.

897

898 Raselimanana AP (2008) Herpétofaune des forêts sèches malgaches. In Goodman SM,
899 Wilmé L (eds). *Les forêts sèches de Madagascar*. *Malagasy Nature* 1: 46–75.

900

901 Rosser AM, Leader-Williams N (2010) Protection or use: a case of nuanced trade-offs? In
902 Leader-Williams N, Adams WM, Smith RJ (eds) *Trade-offs in conservation: deciding what*
903 *to save*. Wiley-Blackwell, Oxford , pp 135–156.

904

905 Safford R, Hawkins F (eds) (2013) *Birds of Africa, vol. VIII: the Malagasy region*.
906 Christopher Helm, London.

907

908 Scales BR, Marsden SJ (2008) Biodiversity in small-scale tropical agroforests: a review of
909 species richness and abundance shifts and the factors influencing them. *Environ Conserv* 35:
910 160–172.

911

912 Scharlemann JPW, Kapos V, Campbell A, Lysenko I, Burgess ND, Hansen MC, Gibbs HK,
913 Dickson B, Miles L (2010) Securing tropical forest carbon: the contribution of protected
914 areas to REDD. *Oryx* 44: 352–357.

915

916 Scott DM, Brown D, Mahood S, Denton B, Silburn A, Rakotondraparany F (2006) The
917 impacts of forest clearance on lizard, small mammal and bird communities in the arid spiny
918 forest, southern Madagascar. *Biol Conserv* 127: 72–87.
919

920 Seddon N, Tobias JA (2007) Population size and habitat associations of the long-tailed
921 ground-roller *Uratelornis chimaera*. *Bird Conserv Int* 17: 1–12.
922 34: 287–304.
923

924 Sekercioglu CH, Loarie SR, Oviedo Brenes F, Ehrlich PR, Daily GC (2007) Persistence of
925 forest birds in the Costa Rican agricultural countryside. *Conserv Biol* 21: 482–494.
926

927 Shafer CL (2015) Cautionary thoughts on IUCN protected area management categories V-VI.
928 *Global Ecol Conserv* 3: 331– 348.
929

930 Smith RJ, Veríssimo D, Isaac NJB, Jones KE (2012) Identifying Cinderella species:
931 uncovering mammals with conservation flagship appeal. *Conserv Lett* 5: 205–212.
932

933 Stattersfield AJ, Crosby MJ, Long AJ, Wege DC (1998) Endemic bird areas of the world:
934 priorities for biodiversity conservation. BirdLife International, Cambridge.
935

936 Tews J, Brose U, Grimm V, Tielbörger K, Wichmann MC, Schwager M, Jeltsch F (2004)
937 Animal species diversity driven by habitat heterogeneity/diversity: the importance of
938 keystone structures. *J Biogeogr* 31: 79–92.
939

940 Theisinger O, Ratianarivo MC (2015) Patterns of reptile diversity loss in response to
941 degradation in the spiny forest of southern Madagascar. *Herpetol Conserv Biol* 10: 273–283.
942

943 Tilman D, May RM, Lehman CL, Nowak MA (1994) Habitat destruction and the extinction
944 debt. *Nature* 371: 65–66.
945

946 Tucker CM, Cadotte MW, Davies TJ, Rebelo TG (2012) Incorporating geographical and
947 evolutionary rarity into conservation prioritisation. *Conserv Biol* 26: 593–601.
948

949 Vallan D (2002) Effects of anthropogenic environmental changes on amphibian diversity in
950 the rain forests of eastern Madagascar. *J Trop Ecol* 18: 725–742.
951

952 Venter O, Fuller RA, Segan DB, Cawardine J, Brooks T, Butchart SHM, Di Marco M,
953 Iwamura T, Joseph L, O’Grady D, et al. (2014) Targeting global protected area expansion for
954 imperilled biodiversity. *PLoS Biol* 12: e1001891
955

956 Virah-Sawmy M, Gardner CJ, Ratsifandrihamanana AN (2014) The Durban vision in
957 practice: experiences in participatory governance of Madagascar’s new protected areas. In
958 Scales IR (ed) *Conservation and environmental management in Madagascar*. Routledge,
959 London, pp. 216–252.
960

961 Visconti P, Bakkenes M, Smith , Joppa L, Sykes RE (2015) Socio-economic and ecological
962 impacts of global protected area expansion plans. *Phil Trans R Soc B* 370: 20140284.
963

964 Watson JEM, Dudley N, Segan DB, Hockings M (2014) The performance and potential of
965 protected areas. *Nature* 515: 67–73.
966

967 Watson JEM, Darling ES, Venter O, Maron M, Walston , Possingham HP, Dudley S,
968 Hockings, Barnes M, Brooks TM (2016) Bolder science needed now for protected areas.
969 *Conserv Biol* 30: 243–248.
970

971 Wilmé L (1996) Composition and characteristics of bird communities in Madagascar. In
972 Lourenço WR (ed), *Biogéographie de Madagascar*. Editions ORSTOM, Paris, pp 349–362.
973

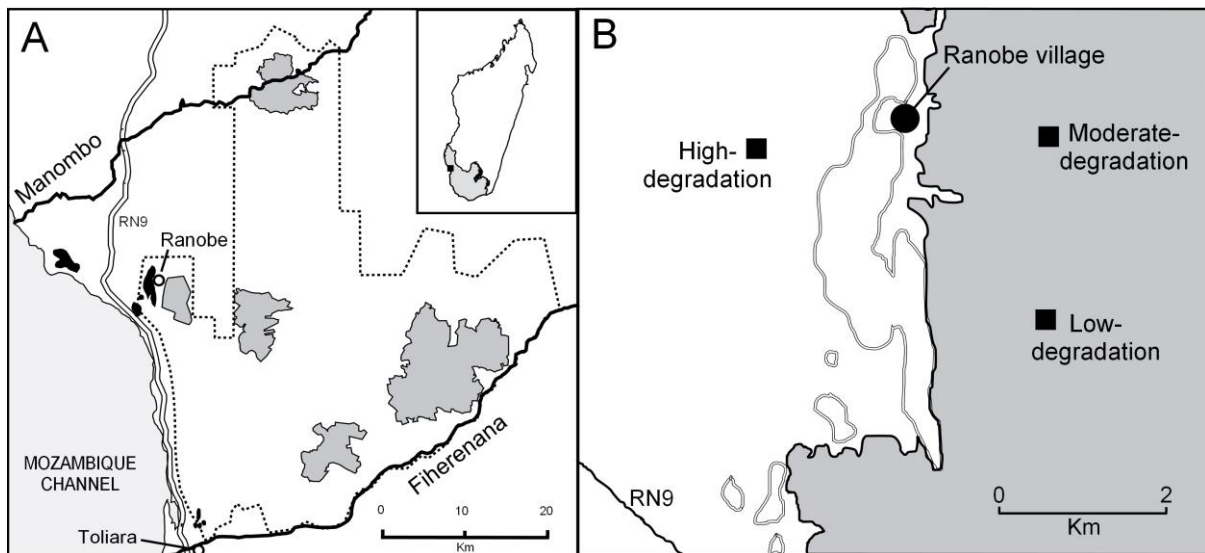
974 WWF (2010) Plan d’aménagement et de gestion de PK32-Ranobe. WWF Madagascar and
975 Western Indian Ocean Programme Office, Antananarivo.
976

977 Zimmerer KS, Galt RE, Buck MV (2004) Globalization and multi-spatial trends in the
978 coverage of protected-area conservation (1980–2000). *Ambio* 33: 520–529.
979

980 **Figure Legends**

981

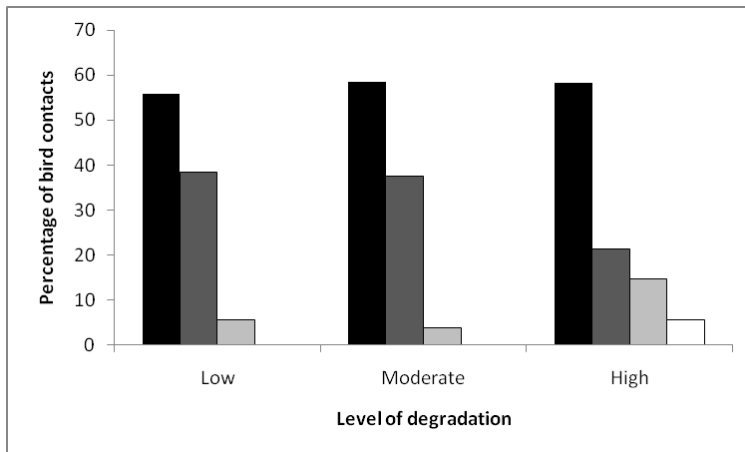
982 **Fig. 1** Map of: A) Ranobe PK32 protected area (dotted line) showing location of five strict
983 conservation zones (grey shading), wetlands and rivers (black shading/lines) and Ranobe
984 village; and, B) location of three vegetation treatments used to survey bird and reptile
985 communities across a gradient of degradation (forest cover, grey shading; wetlands, double
986 line). Inset shows location of Ranobe PK32 within Madagascar (black square) and limits of
987 spiny forest ecoregion following Goodman and Raherilalao (2013) (grey shading)



988

989

990 **Fig. 2** Endemism status of birds at Ranobe expressed as a percentage of contacts from 48
991 point counts at three sites across a gradient of degradation. Black, Madagascar endemic; dark
992 grey, regional endemic; light grey, indigenous non-endemic; white, introduced. Regional
993 endemic species are defined as restricted to Madagascar and the western Indian Ocean islands
994 (Comoros, Mascarene and Seychelles archipelagos)



995

996 **Table 1** Disturbance history and vegetation description of three habitat treatments used to
 997 investigate the impacts of degradation on birds and reptiles at Ranobe, southwest
 998 Madagascar.

999

Treatment	Disturbance history	Habitat description
Low disturbance (<i>Low</i>)	Low level charcoal production since 2007	Relatively closed canopy dominated by <i>Didierea madagascariensis</i> and hardwood trees, with no understory shrub layer. Some charcoal production resulting in small openings, but canopy generally unbroken. Thick leaf litter layer.
Moderate disturbance (<i>Mod</i>)	Intensive charcoal production since 1995	Broken canopy dominated by <i>Didierea madagascariensis</i> , with hardwood trees largely absent. Small openings are frequent and possess a dense shrub layer of regenerating stumps. Characterised by piles of dead branches and bark left over from charcoal production. Thin leaf litter layer.
High disturbance (<i>High</i>)	Forest cleared for shifting cultivation in 2001, regenerating naturally since 2004/5	Dense shrub layer (height of 1-2m) of regenerating stumps dominated by <i>Cedrelopsis grevei</i> and <i>Fernandoa madagascariensis</i> , with no litter layer. Relict individual trees and small forest patches (< 1ha) occur within a mosaic pattern.

1000

1001

1002 **Table 2** Scoring criteria for Conservation Value Index (CVI) attributes, used to quantify the conservation value of individual bird and reptile
 1003 species at Ranobe, southwest Madagascar. EBA = Endemic Bird Area (Stattersfield *et al.* 1998), PA = protected area.

1004

Taxonomic group	Score	Geographic scale of endemism (G)	Taxonomic level of endemism (E)	Representation in sample PAs (R)	Hunting/collection pressure (C)	Degradation tolerance (T)
Birds	1	Indigenous, non-endemic species	Indigenous, non-endemic species	Recorded in 12-14 PAs (n > 85%)	N/A	Tolerant of modified or artificial habitats
	2	Endemic to western Indian Ocean	Endemic species	Recorded in 8-11 PAs (55 > n < 85%)	N/A	N/A
	3	Widespread Madagascar endemic	Endemic genus	Recorded in 4-7 PAs (30 > n < 50%)	N/A	Tolerant of edge effects, medium-intensity degradation or secondary growth.
	4	Endemic to dry regions of Madagascar	Endemic subfamily	Recorded in 2-3 PAs (10 > n < 20%)	N/A	N/A
	5	EBA species	Endemic family	Recorded in only 1 PA (n < 10%)	N/A	Intolerant of low-intensity degradation
Reptiles	1	Indigenous, non-endemic species	N/A	Recorded in 12-15 PAs (n > 75%)	No known threat	Tolerant of modified or artificial habitats
	2	Widespread endemic, occurring in dry and humid regions	N/A	Recorded in 8-11 PAs (45 > n < 75%)	N/A	N/A
	3	Endemic to dry regions	N/A	Recorded in 4-7 PAs (20 > n < 45%)	Known threat (CITES Appendix I and II), but not known to cause local extirpations	Tolerant of edge effects, medium-intensity degradation or secondary growth.
	4	Endemic to one bioclimatic region ^a	N/A	Recorded in 2-3 PAs (10 > n < 20%)	N/A	N/A
	5	Local endemic, range size estimated as < 10,000 km ²	N/A	Recorded in only 1 PA (n < 10%)	Threat known to have caused local extirpations or severe population declines	Intolerant of low-intensity degradation

1005 ^aFollowing Cornet 1974

1006 **Table 3** Bird species recorded at Ranobe showing attributes used in Conservation Value
 1007 Index (CVI) and frequency-weighted CVI scores for three sites across a gradient of
 1008 degradation: Low, Mod and High indicate low-, moderate- and high-degradation treatments.
 1009 CVI attributes: *G* – geographic scale of endemism, *E* – taxonomic level of endemism, *R* –
 1010 representation in sample protected areas, *T* – degradation tolerance. Asterisks indicate species
 1011 endemic to the spiny forest Endemic Bird Area (Stattersfield *et al.* 1998).

Species	CVI attribute scores				CVI score	Frequency-weighted CVI		
	G	E	R	T		Low	Mod	High
* <i>Monias benschi</i>	5	5	4	5	90	90	22.5	22.5
* <i>Xenopirostris xenopirostris</i>	5	5	3	5	80	0	11.4	80
* <i>Coua cursor</i>	5	4	3	5	72	72	20.6	30.9
* <i>Uratelornis chimaera</i>	5	5	4	3	70	0	0	0
* <i>Thamnornis chloropetoides</i>	5	5	2	5	70	70	60.0	0
* <i>Newtonia archboldi</i>	5	5	2	5	70	70	47.6	22.4
<i>Coua ruficeps olivaceiceps</i>	4	4	2	5	56	40.0	56	0
<i>Calicalicus madagascariensis</i>	3	5	2	5	56	56	56	0
<i>Artamella viridis</i>	3	5	2	5	56	32.0	56	0
<i>Vanga curvirostris</i>	3	5	1	5	48	48	29.2	4.2
<i>Coua cristata</i>	3	4	1	5	42	42	36.6	25.7
<i>Falco zoniventris</i>	3	2	3	5	40	0	40	0
<i>Falcula palliata</i>	4	5	1	3	36	36	36	0
<i>Leptosomus discolor</i>	2	5	2	3	35	0	0	0
* <i>Nesillas lantzii</i>	5	2	2	3	35	35	0	11.7
<i>Newtonia brunneicauda</i>	3	5	1	3	32	32	30.7	14
<i>Leptopterus chabert</i>	3	5	1	3	32	19.2	16	32
<i>Aviceda madagascariensis</i>	3	2	3	3	30	0	30	0
<i>Neomixis striatigula</i>	3	3	2	3	30	21.5	30	10.8
<i>Cuculus rochii</i>	3	2	2	3	25	25	11.7	5
<i>Polyboroides radiatus</i>	3	2	1	3	20	20	10	0
<i>Buteo brachypterus</i>	3	2	1	3	20	0	20	0
<i>Mirafra hova</i>	3	2	3	1	20	0	0	20
<i>Copsychus albospecularis</i>	3	2	1	3	20	20	19.3	14.3
<i>Treron australis</i>	2	1	2	3	15	0	0	15
<i>Nectarinia notata</i>	2	1	2	3	15	0	15	3.75
<i>Ploceus sakalava</i>	4	1	2	1	15	0.7	15	5.0
<i>Accipiter francesiae</i>	2	1	1	3	12	0	12	0
<i>Turnix nigricollis</i>	2	1	1	3	12	2.1	0	12
<i>Nesoenas picturata</i>	2	1	1	3	12	9.7	12	2.9
<i>Coracopsis vasa</i>	2	1	1	3	12	12	12	0
<i>Coracopsis nigra</i>	2	1	1	3	12	5.0	12	0
<i>Phedina borbonica</i>	2	1	3	1	12	0	12	12
<i>Hirundo rustica</i>	1	1	5	1	12	0	0	0
<i>Hypsipetes madagascariensis</i>	2	1	1	3	12	12	2.6	6.8
<i>Terpsiphone mutata</i>	2	1	1	3	12	9.7	12	4.6
<i>Neomixis tenella</i>	3	3	1	1	12	12	11.7	9.3
<i>Cisticola cherina</i>	2	1	3	1	12	0	0	12
<i>Nectarinia souimanga</i>	2	1	1	3	12	11.5	12	8.8
<i>Dicrurus forficatus</i>	2	1	1	3	12	12	9	9.5
<i>Falco peregrinus</i>	1	1	4	1	10	0	0	10
<i>Agapornis canus</i>	3	2	1	1	10	3.3	3.3	10
<i>Tachymarptis melba</i>	1	1	4	1	10	0	10	0
<i>Eurystomus glaucurus</i>	1	1	2	3	10	0	10	0

<i>Upupa marginata</i>	3	2	1	1	10	10	6.4	8.6
<i>Falco newtoni</i>	2	1	2	1	9	4.1	3.3	9
<i>Caprimulgus madagascariensis</i>	2	1	2	1	9	9	0	0
<i>Falco concolor</i>	1	1	3	1	8	0	0	8
<i>Foudia madagascariensis</i>	3	1	1	1	8	0.2	0.8	8
<i>Milvus migrans</i>	1	1	2	1	6	0	0	0
<i>Oena capensis</i>	1	1	2	1	6	3.4	1.4	6
<i>Centropus toulou</i>	2	1	1	1	6	4.3	4.3	6
<i>Apus barbatus</i>	1	1	2	1	6	3	6	0
<i>Merops superciliosus</i>	1	1	2	1	6	3.7	3.3	6
<i>Corvus albus</i>	1	1	2	1	6	0	0	6
<i>Numida meleagris</i>	1	1	2	1	6	0	0	6
Total conservation value of treatment						856.4	825.7	478.6

1012

1013

1014 **Table 4** Reptile species recorded at Ranobe showing attributes used in Conservation Value
 1015 Index (CVI) score and relative frequency-weighted CVI scores for three sites across a
 1016 gradient of degradation: Low, Mod and High indicate low-, moderate- and high-degradation
 1017 treatments. CVI attributes: *G* – geographic scale of endemism, *R* – representation in sample
 1018 protected areas, *C* – collection/hunting threat, *T* – degradation tolerance. Locally-endemic
 1019 species are indicated by an asterisk.

1020

Species	CVI attribute scores				CVI score	Frequency-weighted CVI		
	G	R	C	T		Low	Mod	High
<i>Pyxis arachnoides</i>	4	3	5	5	70	70	0	0
* <i>Voeltzkowia petiti</i>	5	4	1	5	54	14.7	54	0
* <i>Tracheloptychus petersi</i>	5	4	1	5	54	22.1	54	2.5
<i>Geckolepis polylepis</i>	4	4	1	5	48	48	32.8	0
<i>Paroedura androyensis</i>	4	3	1	5	42	14	42	0
* <i>Pygomeles braconnieri</i>	5	4	1	3	36	14.4	36	0
<i>Voeltzkowia rubrocaudata</i>	3	3	1	5	36	36	0	10.3
* <i>Zonosaurus quadrilineatus</i>	5	4	1	3	36	36	32	8
<i>Ithycyphus oursi</i>	3	3	1	5	36	0	36	0
* <i>Liophidium chabaudi</i>	5	4	1	3	36	36	28.8	21.6
<i>Madascincus igneocaudatus</i>	3	2	1	5	30	12	30	0
<i>Madagascarophis ocellatus</i>	4	3	1	3	28	28	0	0
<i>Blaesodactylus sakalava</i>	3	1	1	5	24	24	16	0
<i>Zonosaurus karsteni</i>	3	3	1	3	24	24	24	0
<i>Madagascarophis meridionalis</i>	3	3	1	3	24	0	24	0
<i>Trachylepis aureopunctata</i>	3	2	1	3	20	6.7	20	3.3
<i>Heteroliodon occipitalis</i>	3	2	1	3	20	10	20	0
<i>Leioheterodon geayi</i>	3	2	1	3	20	20	0	0
<i>Typhlops arenarius</i>	3	2	1	3	20	0	6.2	20
<i>Typhlops decorsei</i>	3	2	1	3	20	0	20	0
<i>Lygodactylus verticillatus</i>	4	4	1	1	16	16	4	0
<i>Phelsuma mutabilis</i>	3	1	3	1	16	16	10.3	2.3
<i>Amphiglossus ornaticeps</i>	2	2	1	3	16	9.6	16	0
<i>Oplurus cyclurus</i>	2	2	1	3	16	16	6.5	0.73
<i>Lygodactylus tuberosus</i>	4	3	1	1	14	0	0	14
<i>Paroedura picta</i>	3	2	1	1	10	3.8	1.9	10
<i>Furcifer verrucosus</i>	3	2	1	1	10	10	2.9	0
<i>Chalarodon madagascariensis</i>	2	2	1	1	8	8	4	3.0
<i>Trachylepis elegans</i>	2	1	1	1	6	4.9	6	2.9
<i>Dromicodryas bernieri</i>	2	1	1	1	6	6	6	6
<i>Mimophis mahfalensis</i>	2	1	1	1	6	4.5	6	5.3
<i>Hemidactylus mercatorius</i>	1	1	1	1	4	4	3.4	2.9
Total conservation value of treatment						514.7	542.8	112.7

1021
1022

1023
1024
1025
1026
1027
1028

Table 5 Observed and estimated species richness and Conservation Value Index (CVI) score for birds and reptiles at three sites across a gradient of disturbance at Ranobe, southwest Madagascar.

	Low disturbance	Moderate disturbance	High disturbance
Observed bird richness	36	43	37
Estimated bird richness	42.0	46.8	39.7
Bird CVI	856.4	825.7	478.6
Observed reptile richness	27	27	15
Estimated reptile richness	30.5	34.2	19.1
Reptile CVI	514.7	542.8	112.7