

1 **Effect of forest landscapes composition and configuration on bird community and its**
2 **functional traits in a hotspot of biodiversity of Chile**

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28

29 **Abstract**

30 Understanding the effect of landscape configuration on the bird species richness and their
31 functional traits (dietary preferences) is important to link the conservation and restoration plans
32 to the production of the crops. Our aims were: 1) to study the influence of forest types (native,
33 mixed and plantations) on the bird species richness in two agroforestry landscapes
34 (heterogeneous/homogeneous); 2) to assess the effect of size/density of forest patches in the
35 birds' functional traits; 3) to evaluate the effect of isolated trees on them, and 4) to discuss
36 conservation and restoration measures for the birds' functional traits in agroforestry landscapes.
37 We used hierarchical occupancy models to evaluate the effect of different landscape metrics and
38 detectability measures on bird communities. We recorded a total of 64 bird species. The
39 estimated species richness was considerable higher in homogeneous landscape (31.7 ± 2.7) than
40 heterogeneous (27.3 ± 2.5). Our results showed the bird assemblage had a positive trend with
41 native forests, negative with mixed forests and neutral trend for plantations. The granivores and
42 insectivore's species showed significant preferences for homogeneous landscape, while
43 omnivores had significant preferences for heterogeneous landscape. Carnivores/Piscivores and
44 herbivores/frugivores species did not show preferences by any landscape type. The response of
45 functional traits depended on different forests attributes. The isolated trees had a significant
46 effect on the birds' functional traits. In conclusion, it is necessary a deep knowledge about the
47 relationship between the landscape configuration and the bird species richness/functional traits.
48 These findings could help in the future with the conservation, restoration, and rewilding policies
49 in this important hotspot of biodiversity, avoiding alterations in the ecosystem services.

50 **Keywords:** Agroforestry; Bird guilds; Landscape metrics; Occupancy models; Plantations.

51

52 **Introduction**

53 The composition (land-use percentage) and disposition of elements that belong to the
54 agroforestry landscapes have serious implications in the distribution of the species and their
55 richness (Amos et al., 2013). The fragmentation of forest areas as a result of the land-use changes
56 has modified the configuration of the elements permanently, and consequently the wildlife must
57 adapt to the new situation. However, this usually supposes a declination or loss in their
58 populations (Batáry et al., 2020; Carvajal et al., 2018; Daskalova et al., 2020; Docherty et al.,
59 2020; García-Navas et al., 2020; Santos et al., 2021; Traba & Morales, 2019). The fragmentation
60 process is not uniform and can create different structures within the landscape configuration such
61 as isolated trees, tree lines, corridors, large and homogeneous patches, smaller patches, etc.
62 (Altamirano et al., 2020; Daskalova et al., 2020; Martin et al., 2020; Valente & Betts, 2019).
63 This way, understanding how the distribution and location of these elements affect wildlife
64 (richness and functional traits) will help in the land-sharing/sparing debate (Arroyo-Rodríguez
65 et al., 2020; Brambilla, 2019; Cannon et al., 2019; Finch et al., 2019; Gilroy et al., 2014; Lamb
66 et al., 2019), as well as to design wildlife friendly areas in a global rewilding context (Kirk et al.,
67 2020; Perino et al., 2019).

68 This landscape configuration can increase the mobility between different habitats (Väli
69 et al., 2020; Whytock et al., 2018), and therefore, it promotes the presence of many species with
70 different functions (Gardiner et al., 2019). The functional traits can affect significantly some of
71 the ecosystem services provided by animals in agroecosystems such as seed dispersion,
72 pollination, pest suppression (Arroyo-Rodríguez et al., 2020; Barros et al., 2019; Benedetti et al.,
73 2020; Klingbeil & Willig, 2016). Therefore, it is necessary to understand the relationship
74 between landscape elements and functional traits, because it is the keystone to develop
75 conservation and restoration plans (Benedetti et al., 2020; Dekeukeleire et al., 2019; Harris et
76 al., 2020).

77 Birds have many functions within the ecosystems (Barbaro et al. 2017; Redlich et al.
78 2018; Acosta-Rojas et al. 2019; Benedetti et al. 2020; Muñoz-Sáez et al. 2020), but currently
79 there is a strong decline in some of their populations (Bowler et al., 2019). Nevertheless, several
80 studies showed that the landscape configuration affected the presence/absence of the species
81 and/or communities (Barros et al., 2019; Donald et al., 2019; Halstead et al., 2019; Ibarra &
82 Martin, 2015; Morante-Filho et al., 2020; Neilan et al., 2019). The absence of some elements
83 such as isolated trees or tree lines can disrupt the mobility of these species, and consequently
84 some suitable patches are not available for these species because they cannot get access to them
85 (Akresh et al., 2021). Also, these same elements avoid the species being perceptible for their
86 natural predators. On the other hand, the forest patches with different typologies and/or
87 vegetation communities are essential for birds, since they provide roosts, breeding places, prey,
88 etc. (Jiménez-Franco et al., 2018). Indeed, roads can increase noise contamination and affect the
89 presence of some species (Cooke et al., 2020; Hall et al., 2016; Sol et al., 2020).

90 La Araucania region is placed in the centre-south of Chile and is part of a biodiversity
91 hotspot (Fuentes-Castillo et al., 2020; Myers et al., 2000). However, in the last decades, the
92 central valley has suffered large changes in landscape composition based on anthropogenic
93 transformations with a great loss of native forests, where a significant proportion (21% between
94 1970-1990; 17% between 1990-2000; and 22% between 2000-2010) of native forest became in
95 other land uses (forest plantations of non-native species, shrublands, agriculture and pasture;
96 Miranda et al. 2015, 2017). Also, the presence of fires is high, especially due to human activity
97 (Miranda et al., 2020). However, this transformation is not uniform and we can basically find
98 two types of landscape: heterogeneous landscapes with small patches of crops which are
99 interconnected. and homogeneous, based on areas with large crop extensions (see Figure 1).

100 Our hypothesis was that species richness and functional traits of the bird community will
101 be higher in those landscapes with a heterogeneous configuration and with a high availability of

102 landscape elements such as native forest (Atauri & de Lucio, 2001). This way, the aims of this
103 study were: 1) to study and assess the influence of forest patches types on the bird species
104 richness and their functional traits in both landscape types (homogeneous vs. heterogeneous); 2)
105 to evaluate the effect of isolated trees on birds' functional traits; and 3) to discuss conservation
106 and restoration measures for functional bird diversity in agroforestry landscapes. We hope that
107 this study helps wildlife managers to understand landscape composition and configuration in
108 relation to species richness in order to conserve important hotspot of biodiversity.

109 **Material and Methods**

110 *Study area*

111 The study area is located in the central valley of La Araucania region in the centre-south of
112 Chile (mid-coordinates are 38°51'S latitude, 72°20'W longitude; Figure 1). The valley is
113 dominated by alluvial plains where the native forest has disappeared in favour of intensive crops,
114 urban areas and forest plantations (Miranda et al., 2015, 2017). Today, most of the land cover
115 (ca.75%) in the Araucania Central Valley is crop and pasture land, with a recent increase in
116 exotic tree plantations (ca. 11%; Miranda et al. 2015). In this area we can recognise two large
117 landscape types (Rey Benayas et al., 2020): one is composed by relatively small crops with a
118 dense network of reticulate forest which connects small forest patches (hereafter, Landscape A)
119 and other composed by relatively large crop patches (homogeneous monoculture) with a sparse
120 network of reticulate forest and scarce forest patches (hereafter, Landscape B; Figure 1).

121 *Birds community monitoring*

122 To monitor the bird assemblage, we selected eight roads/tracks (secondary roads, rural
123 ways), inside of our two landscape types, which were suitable to drive on them (without fences
124 or doors), long enough (>10 km) and as straight as possible. Then, we randomly selected four of
125 them for each landscape type (Figure 1). For each transect, we performed 10 count points

126 separated by 1 km between them (n = 80 count points) and located at approx. 50 m away from
127 the road/tracks. To finish the transect on time, before it was too dark, we commuted by car
128 between points. We waited 5 minutes to start the count point after turning off the car. A minimum
129 of two observers participated in every survey. Each week, we recorded one transect for each
130 landscape type between November (2018) and February (2019) (austral spring and summer). In
131 total, we replicated each point 5 times (n = 400 surveys).

132 The transects started 2 hours before the sunset and each point was observed for 10 minutes
133 (4000 minutes in total). We recorded the bird species present and individuals number observed
134 through visual (8 x 42 binoculars) and auditory records within 50 metres of the point. The order
135 of the census points (start and end) was inverted monthly to avoid bias caused by the starting
136 hour. This way, if the order of a transect was from point 1 to point 10 one month, the next month
137 it would be from point 10 to point 1. The transects were always made under the suitable
138 meteorological conditions for a correct visualization, making sure there was no rain or strong
139 wind to avoid misidentification (Zuberogoitia et al., 2020). We counted for each point the
140 presence/absence of each bird species. Although usually bird monitoring surveys are made in the
141 sunrise, we choose the sunset because this way could also record some birds with nocturnal
142 habits (e.g., *Tyto alba* and *Asio flammeus*). Some studies show that the peak of detectability of
143 bird species is not related to the sunrise (Gordo et al., 2021) and there are not differences between
144 the hours of the survey (de Araújo et al., 2021). We approach this circumstance into our models
145 using the variable "minutes before the sunset" to model the detection probability.

146 Each bird species was assigned a functional trait based on the main diet of the species
147 (www.avesdechile.cl; Supplementary Material 1). Birds feeding behaviour is fundamental to
148 understand the response of species to environmental changes in the ecosystem (Coelho et al.,
149 2016; Ibarra & Martin, 2015; Luck et al., 2013; Mahoney et al., 2019). Therefore, each species

150 was categorized in the following groups: 1) Granivore; 2) Insectivore; 3) Omnivore; 4)
151 Carnivores/Piscivores, and 5) Herbivores/Frugivores.

152 *Landscape and temporal variables*

153 For each survey point, we delimited a buffer area of 500 meters of radius to identify
154 landscape variables (Figure 1). Using Google Earth® imagery (2019) and QGIS software (2004-
155 2016), we draw all the habitat types and structures as vectorial shapes and then transformed them
156 into a raster map with a pixel size of 2 meters. We classified the land-use in the following types:
157 1) farmland; 2) tree lines; 3) mixed forest; 4) native forest; 5) wetland; 6) plantation forest; 7)
158 river; 8) road; and 9) urban. The native forest was composed mainly by Siempreverde and Roble-
159 Raúlí-Coihue forest types (Rey Benayas et al., 2020), where the dominant species are:
160 *Nothofagus obliqua*, *N. dombeyi*, *Myrceugenia exsucca*, *Blepharocalyx cruckshanksii*, *Drimys*
161 *winteri*, *Chisquea coleu*, *Laurelia sempervires*, *Maytenus boaria* and *Persea lingue*. The mixed
162 forest contains the same typical species of the native forest but also non-native species such as
163 *Salix babylonica*, *Eucalyptus globulus*, *Acacia dealbata*, *Populus nigra* and *Ulex europaeus*. The
164 plantations are monoculture areas of *Eucalyptus globulus* and *Pinus radiata*. The tree lines are
165 composed by non-native species such as *Populus alba*. Then, we used the software Fragstats
166 version 4.2. to calculate the following variables of each land-use cover as measures of landscape
167 structure to be related to biodiversity inside the buffer area: 1) total occupied area (ha); 2)
168 percentage of occupied area; 3) patch density (patches number per each 100 ha); and 4) average
169 patch size (ha) (see Supplementary Material 2).

170 For every survey, we recorded the following variables for each point: 1) Minutes before the
171 sunset (min); 2) Temperature (°C); 3) Rainfall of the previous day (mm); and 4) Wind velocity
172 (km/h) (see Supplementary Material 3). These meteorological data were obtained from two sites
173 (Fundo de Maquehue and El Taplón), which are close (< 5 km) to the transect and belong to

174 “Dirección de Meteorología de Chile (meteochile.gob.cl)” and “Red Agrometeorológica del
175 INIA (agromet.inia.cl)”, respectively.

176 *Data analysis*

177 For our analysis, we considered that our sampling unit is the survey point while the transect
178 represents a replica of the landscape. Occupancy models are important to understand the changes
179 in the trends of various species (Pillay et al., 2019; Southwell et al., 2019). Multispecies
180 occupancy models are a more complex framework, aimed at estimating total community richness
181 accounting for differences in species detectability, and few studies have evaluated the effects of
182 environmental variables on bird communities (Kéry & Royle, 2008; Kéry & Schmidt, 2008;
183 Zamora-Marín et al., 2021; Zipkin et al., 2009, 2010). We performed a Bayesian multispecies
184 occupancy model, which is an extension of the single species site occupancy model (MacKenzie
185 et al., 2002), whereby the hierarchical structure combines community and species level attributes
186 within a single analytical framework (Zipkin et al., 2009). The hierarchical models are composed
187 by the ecological process (governed by occupancy probability) and the observational process
188 (governed by detectability probability). Data are compiled as a 2x2 matrix Y with i rows by k
189 columns, corresponding to count points and species, respectively. The number of spatial
190 replicates j for each count point i where the species k was observed is quantified in the matrix Y .
191 The ecological process assumes that count point specific occupancy (i.e. “true”
192 presence/absence) for species $k = 1, 2, \dots, N$ count point, denoted as $z(i, k)$, where $z(i, k) = 1$ if
193 species k occurs in count point i and is zero otherwise.

194 The model for occurrence is specified as $z(i, k) \sim \text{Bernoulli}(\psi_{i,k})$ where $(\psi_{i,k})$ is the probability
195 that species k occurs at count point i . The true occurrence is imperfectly observed, and we define
196 the detection model for species k at count point i in replicate j as $Y(i, k) \sim \text{Binomial}(p_{ik} * z(i, k))$,
197 where p_{ik} is the detection probability of species k for the j^{th} spatial replicate at count point i , given
198 that species k is in fact present at count point i , and $J(i)$ is the number of spatial replicates (i.e.,

199 transects) in cell (count point) i . We modelled the occurrence probability for species k at count
200 point i by incorporating site-specific landscape characteristics (Jiménez-Franco et al., 2019;
201 Zipkin et al., 2009). The model included a linear effect of percentage of each forest type (native,
202 mixed forest and plantations) in the buffer area and the qualitative variable of landscape type
203 (homogeneous/heterogeneous). We also include effects of functional traits (dietary preferences:
204 granivores, insectivores, omnivores, carnivores/piscivores and herbivores/frugivores) in
205 occupancy probability for species k as a qualitative covariate. All quantitative habitat covariables
206 were standardized by subtracting the mean and dividing by the standard deviation. We assumed
207 that detection probabilities varied depending on the species but were not influenced by the survey
208 characteristics: $\text{logit}(p_k) \sim \text{Normal}(\mu_{lp,k}, \sigma^2_{lp,k})$. We added an additional hierarchical component of
209 the model by assuming that the species-level parameters were random effects, each governed by
210 community-level “hyper-parameters”. For example, a community response (mean across
211 species) for a count point and standard deviation (among species) were estimated, so that the
212 hyper-parameters are simply the mean and variance for each covariate, as measured across
213 species (Kéry & Royle, 2016). Model was fitted using *JAGS* (Plummer, 2003), run in R. 3.6.2
214 (R Core Team, 2016) with the package *jagsUI* (Kellner, 2015), using uninformative priors, three
215 chains, 15000 iterations and a burn-in of 5000 iterations and a thin rate of 2.

216 We used single season occupancy model to estimate probability of habitat occupancy (ψ) at
217 functional trait level (granivores, insectivores, omnivores, carnivores/piscivores, and
218 herbivores/frugivores) while accounting for imperfect detection (MacKenzie et al., 2002). First,
219 we fit a set of detection probability models (p). Using the best detection models, we developed
220 a set of occupancy probability models for each functional trait. Detection probability and
221 occupancy probability models were fitted according to all possible additive combinations of
222 temporal and landscape variables as covariates, respectively. We also include the type of
223 landscape (A or B) as a possible additive or interaction covariate in both modelling processes.

224 We tested the landscape variables of patch density and average patch size for the different forest
225 covers types and isolated trees and the total occupied area for total farmland. Tested covariates
226 did not show a correlation between them (Pearson < 0.6), therefore there cannot be
227 multicollinearity in the models. All continuous covariates were standardized before the
228 modelling process to adequately compare the parameters estimated by the models. Models were
229 fitted using *occu* function of the *unmarked* package in R software (Fiske & Chandler, 2011).
230 Akaike's Information Criterion (AIC) was employed to compare candidate models, and to
231 estimate parameter, unconditional standard errors and 95% confidence via model averaging
232 (Burnham & Anderson, 2002). Model averaging was done using *AICcmodavg* package in R
233 software (Mazerolle, 2020). Candidate models that represented a 90% confidence set of best-
234 ranked regression models were averaged (i.e., models which cumulative Akaike weight ≤ 0.9 ;
235 Table 1).

236 Medians of estimated occupancy probabilities using observed landscape variables as
237 predictors were compared between landscape type (A or B) for each functional trait group through
238 Mann-Whitney nonparametric statistical test. We did not use means comparison tests because
239 our data did not present a normal distribution according to Shapiro-Wilk test (p -values equal to
240 1.66×10^{-7} , 9.71×10^{-8} , 0.01, 2.02×10^{-10} and 2.48×10^{-3} for granivores, insectivores, omnivores,
241 carnivores/piscivores and herbivores/frugivores, respectively). Only medians for granivores,
242 insectivores and omnivores were compared, since there was equality of variance between
243 landscape types just for these groups according to Levene's test (p -values equal to 0.36, 0.52,
244 0.41, 1.59×10^{-3} and 4.70×10^{-5} for granivores, insectivores, omnivores, carnivores/piscivores and
245 herbivores/frugivores, respectively). It was not possible to perform a multiple comparison
246 analysis between granivores, insectivores and omnivores, because there was not equality of
247 variance between these groups also according to Levene's test (p -value equal to 1.53×10^{-11}). All
248 statistical tests were performed using a 95% confidence level.

249 We used the parametric bootstrap approach to obtain p -values from sums of squares, Chi-
250 square and Freeman-Tukey fit statistics that quantified the fit of a model to a data set, and as a
251 measure of the goodness of fit of the worst model used in the model averaging process for each
252 functional group (Table 1). We simulated 10,000 bootstrap samples for each fit assessment
253 (Supplementary Material 4). We only evaluated these worst models because the number of totals
254 fitted models was very high (due to the large number of possible covariate combinations). If the
255 results suggest that the models provided an adequate fit to the data, then the rest of the models
256 used in the averaging model probably also provided an adequate fit to the data, because they
257 have a better AIC value. A dispersion parameter (\hat{c}) was calculated as the ratio of the observed
258 Chi-square fit statistic to the mean of the simulated distribution.

259 **Results**

260 *Effects of landscape structure on bird assemblage and species richness*

261 We recorded a total of 64 bird species in both landscapes (Supplementary Material 1). Our
262 models showed that the average number of estimated species per site is 29.7 ± 3.5 (Mean \pm SD),
263 with values higher than observed (20.0 ± 3.5). The point with fewer species had 12 species, while
264 there were two points with a maximum of 27 species. In landscape A, we recorded 46 species
265 (71.9%) while landscape B had a total of 61 species (95.3%; Supplementary Material 1).
266 Although the landscape B had a higher species number, we did not find differences between both
267 landscape types for the observed species richness (19.2 ± 3.4 and 21.6 ± 3.3 , landscape A and B
268 respectively; $W = 534$, $p < 0.05$; Figure 2), but, the estimated species richness was considerably
269 higher in landscape B (31.7 ± 2.7) than landscape A (27.3 ± 2.5 ; Figure 2). Twelve species were
270 only observed in a single point and had a very low naïve occupancy probability (0.012), while
271 the species *Vanellus chilensis* was observed in 78 out of the 80 points with the highest naïve
272 occupancy probability highest of all the bird assemblage (0.975; see more details in
273 Supplementary Material 1).

274 Our results showed that the bird assemblage had a positive trend with the native forest
275 (Figure 3a), a negative trend with mixed forest (Figure 3b), and a neutral trend for plantations
276 (Figure 3c), where in all the cases the estimated richness in landscape B was higher than in
277 landscape A. The result of multispecies occupancy models for each species-specific showed that
278 occupancy estimates were higher than the detectability estimates for most of the bird species
279 (Supplementary Material 5).

280 The number of species classified according to their functional trait were the following:
281 granivores (8 species), insectivores (29 species), omnivores (6 species), carnivores/piscivores
282 (14 species) and herbivores/frugivores (7 species). The estimated occupancy probabilities were
283 similar for granivores group, however the insectivorous group had a higher probability of
284 occupation in landscape B rather than in landscape A, while the omnivores group had a higher
285 probability in landscape A. The carnivorous/piscivorous group did not show differences between
286 both landscapes while the herbivorous/frugivorous group had a higher occupancy probability in
287 landscape A (Figure 4).

288 *Effect of landscape composition and configuration in the bird functional traits*

289 Bootstrap p -values for the worst model used in the model averaging process for each
290 functional trait group based on SSE, Chi-square and Freeman-Tukey statistics were 0.46, 0.82,
291 and 0.44 for granivores; 0.42, 0.40 and 0.40 for insectivores; 0.48, 0.40, and 0.44 for omnivores;
292 0.49, 0.40 and 0.45 for carnivores/piscivores; and 0.48, 0.75 and 0.44 for herbivores/frugivores,
293 respectively. These results suggest that the models provided an adequate fit to the data
294 (Supplementary Material 4). The \hat{c} value were 0.98, 1.00, 1.01, 1.01 and 0.97 for granivores,
295 insectivores, omnivores, carnivores/piscivores and herbivores/frugivores, respectively,
296 indicating that there was not under- or over-dispersion. The response of different functional traits
297 for each landscape variable is dependent according to the landscape type analysed. The

298 importance of each variable in the occupancy probability of each functional trait is summarized
299 in Figure 5.

300 The granivorous group showed a negative response to the mixed forest density in both
301 landscapes (Figure 5 and 6). The insectivores had a positive response for native forest density in
302 landscape B (Figure 5 and 6). The omnivorous group showed a negative response for mixed
303 forest and plantation density in both landscapes. However, they showed a positive response for
304 total farmland area (Figure 5 and 6). The carnivores/piscivores only showed significant
305 differences for the native forest average size, where the response is positive (Figure 5 and 6).
306 The herbivores/frugivores did not show significant differences for landscape variables (Figure
307 5).

308 *Effect of isolated trees on the functional traits*

309 The isolated tree density affected significantly to omnivorous groups (Figure 5), where the
310 omnivores showed a positive response in the landscape B (Figure 6). For the
311 herbivores/frugivores, the average size of isolated tree was negative in landscape A (Figure 6).
312 Nevertheless, the other bird functional traits did not show a significant effect provide by the
313 variables associated with the isolated trees.

314 **Discussion**

315 *Effect of landscape structure in bird assemblage and species richness*

316 Forest elements are vital to maintain the agroecosystem structure, being the keystone in the
317 conservation and management of the landscape, as well as to develop restoration and rewilding
318 plans (Perino et al., 2019; Rey Benayas et al., 2020). Some studies estimate that it is necessary
319 to get 40% of this structures type to achieve the biodiversity conservation goals (Arroyo-
320 Rodríguez et al., 2020). However, the response of each faunistic group is different to the
321 configuration of these elements and this could have serious implications in the functional traits

322 or species richness. This issue is especially important in agroecosystems, because in the decision-
323 making usually, we include the increase of an ecosystem service (e.g., pest suppression) as a
324 promise to persuade the farmers in the need to take wildlife-friendly measures (Cannon et al.,
325 2019; Muñoz-Sáez et al., 2020; Perino et al., 2019). This is especially interesting for birds, a
326 threatened group, highly diversified and which fulfils several roles in the ecosystems.

327 Our results show that in contrast to our initial hypothesis, the landscape with relatively larger
328 crop patches and less reticulate forest had a higher bird species richness compared with the
329 heterogeneous habitats. Moreover, our results showed that the number of observed species was
330 quite similar in both landscape types, but the number of estimated species was considerably
331 different. Human activity is probably more intense in landscape A in terms of noise
332 contamination, a factor known to affect the occurrence of numerous bird species (Cooke et al.
333 2020; Sol et al. 2020), especially during the breeding season (León-Ortega et al., 2017). Also,
334 the number of road is higher in this landscape type and the bird assemblage responds negatively
335 to this variable (Cooke et al., 2019, 2020), although contrary to other studies we did not find
336 differences in our models. Some endemic Chilean species are sensible to the fragmentation and
337 loss of native forest areas (Carvajal et al., 2018). Therefore, it was observed that the
338 homogenization and anthropization process in centre-south Chile affected particularly the
339 specialist species which are less frequent, while the generalist species increased (Ibarra et al.,
340 2017; Ibarra & Martin, 2015). Our results show that the native forest density is positive for
341 insectivorous in landscape B but it is negative for omnivorous in landscape A.

342 However, we observed that the forest typologies in La Araucania influence species richness
343 and the occupancy probability of the birds. This way, our results show that the bird species
344 richness had a positive response to the proportion of native forest, a negative response to the
345 mixed forest and a neutral response to plantations. Although the vegetation composition in the
346 mixed forest is native forest alike, the presence of non-native elements (invasive species such as

347 *Ulex europaeus*) had a highly negative impact on bird species richness in comparison to the
348 plantations. These invasive species could be detrimental for some bird requirements, such as
349 feeding availability, roosts and breeding areas (Acosta-Rojas et al., 2019). This way, Ibarra et al.
350 (2017) found that bird richness is closely associated with native forest and it is lower in the mixed
351 forest. Several studies showed that the presence of *Ulex europaeus* was associated with the
352 anthropogenic transformation (Altamirano et al., 2016; Cockle et al., 2019; Martín-Gallego et
353 al., 2020), and the bird community is closely linked to forest composition (Adams & Matthews,
354 2019; Ibarra & Martin, 2015). Also, in these areas where there is mixed forest the human activity
355 is high and there are suitable zones for Chilean mesocarnivores (Gálvez et al., 2018; Moreira-
356 Arce et al., 2016), which could increase the predation risk of eggs and chicks. Indeed, these
357 humanized areas have a great presence of feral cats and dogs (Seymour et al., 2020) and there
358 can also be found invasive birds (Ascensão et al., 2020).

359 In contrast, the plantations are extensive areas with low human intervention while they are
360 not harvested and they are implanted in uninhabited areas. Plantations usually have fences and a
361 restricted access and therefore it is possible that the predator's presence (both natural and exotic)
362 may be low. This way, generalist birds could use these extensive areas to commute between
363 habitats (Gardiner et al., 2019). Unlike in Spain, where it was found that there were significant
364 differences between the plantations and native forest on bird species richness (Goded et al.,
365 2019), however, our results evidence that mixed forest is worse than plantations. These results
366 are related to the ones found in other places, where the functional traits are not related to spatial
367 landscape attributes and therefore, this could affect some ecosystem services such as the pest
368 suppression (Bełcik et al., 2020; Benedetti et al., 2020).

369 *Effect of land-use cover and landscape on bird functional traits*

370 The configuration and the presence of different landscape elements had a significant impact
371 in occupancy probability of different functional traits (Bełcik et al., 2020). The granivorous

372 species show a positive relationship with farmland areas, especially in landscape B. This makes
373 sense since the main crops in our study area are allocated by grains production (wheat, oats, rye,
374 etc.). This group is affected negatively by mixed forest (both landscape type) but not by native
375 forest. In landscape B, the native forest size is positive for the granivorous species, however the
376 plantations density had a negative effect.

377 Insectivorous birds were the most abundant group. They showed a negative relationship in
378 landscape B with crops while having a positive one with mixed forest density in landscape A
379 and with native forest density in landscape B. However, the plantations did not have an effect on
380 this group. In landscape B, the high native forest density could provide roost and an edge effect
381 where they can perch to hunt their prey (Altamirano et al., 2020; Muñoz-Sáez et al., 2017). Also,
382 these forest elements could help to avoid their predators (Martínez-Núñez et al., 2020). This is
383 especially important if we would like to promote these species for pest suppression in farmland
384 (Muñoz-Sáez et al. 2017, 2020; Martínez-Núñez et al. 2020).

385 Similarly, the omnivorous group had a negative relationship with mixed forest and
386 plantations in both landscapes, also the mixed forest size is negative in landscape B. Since they
387 are a group well adapted to different feeding resources, we found that, in general, the presence
388 of forest habitats (any type) had a negative effect on this group. Probably, the forest areas could
389 increase the predation risk of the omnivorous group. In fact, our models show that the native
390 forest size is positive for carnivores/piscivores.

391 The increase or maintenance of forest elements not always contribute to the conservation of
392 the intensive grassland/pasture matrix (Belcik et al., 2020; Duflot et al., 2018; Kameniar et al.,
393 2021). It is necessary to take into account microclimatic changes to understand the specific-
394 species response (Gaüzère et al., 2020; Threadgill et al., 2020), especially when we measure the
395 functional traits (Klingbeil & Willig, 2016; Redlich et al., 2018).

396 Lee and Goodale (2018) found that the amount of non-productive vegetation increased the
397 species richness and the occupancy probability in heterogeneous agroecosystems only during
398 winter. Therefore, the transformation in land-uses in the forest of La Araucania could filter the
399 species present today (Lindenmayer et al., 2019).

400 *Isolated trees and functional traits*

401 In general, none of the functional traits showed any significant effect with the density and/or
402 the size of the isolated trees. Only, the herbivores/frugivores had a negative response in
403 landscape A for the size of the isolated tree and the omnivores showed a positive response in
404 landscape B for the density of isolated tree. Isolated trees can provide resting areas (Altamirano
405 et al., 2017, 2020; Lindenmayer et al., 2019) and suitable micro-habitats for many species (Basile
406 et al., 2020; Cockle et al., 2019). Also, these elements could act as landmarks for various species
407 and help to reduce the "landscape fear" (Gardiner et al., 2019). Our results did not allow us to
408 understand how these landscape elements affect the bird functional traits and probably the
409 response type is specific-species.

410 **Conclusions and management implications**

411 In conclusion, our study shows that it is necessary to know how the different elements of
412 landscape architecture affect bird assemblages and functional traits. Our results showed that the
413 mixed forest, which has an important human influence, can be more negative than the
414 plantations. Anyway, we observed that the conserved native forest always had a positive effect
415 on species richness and functional traits in La Araucania. Our study evidences that the
416 conservation of native forest is the best policy to maintain the ecosystem services provided by
417 birds such as pest suppression and it is necessary to avoid the entrance of non-native elements
418 (Martín-Gallego et al., 2020).

419 On the other hand, our study is an important contribution to address future restoration plans,
420 because we observed that the maintenance and conservation of the patches of native forest are
421 better and more important than developing a complex reticulated forest. However, we
422 acknowledge the advantages in connectivity that provide the presence of the reticulated forest.

423 Restoration and rewilding policies must take into account our results and the temporal scales
424 for these landscape changes to avoid possible alterations in the ecosystem services provide by
425 bird functional trait (Gilroy et al., 2014). We found that the presence of forest areas in
426 agroecosystems must be attended to avoid the presence of non-native vegetable species. It is
427 necessary to incorporate social aspects into the conservation and restoration plans for a correct
428 trade-off between the needs of different functional traits in an anthropogenic landscape
429 (Klingbeil & Willig, 2016) to get successful.

430 The decision-makers should take into account the imperfect detection of species and the
431 hierarchical models that estimate species richness and occupancy accurately to develop win-win
432 strategies (Tarjuelo et al., 2020), as well as to consider the cost and effort of data acquisition,
433 developing survey designs for multi-species techniques (Zipkin et al., 2009).

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439 **Data availability statement**

440 Data available from Figshare Digital Repository:
441 https://figshare.com/articles/dataset/dataBaseAves_csv/13268924

442 **Animal Research (Ethics)**

443 No applicable

444 **Consent to Participate (Ethics)**

445 No applicable

446 **Consent to Publish (Ethics)**

447 No applicable

448 **Plant Reproducibility**

449 No applicable

450 **Clinical Trials Registration**

451 No applicable

452 **Author Contribution**

453 Conceptualization, F.L.; methodology, F.L. and G.C.; validation, F.L. and M.V.J-F.;
454 formal analysis, F.L., G.C., E.T., C.M-O., M.V.J-F.; investigation, F.L., G.C., E.T., C.M-O. and
455 M.V.J-F.; resources, funding and project administration, F.L.; writing—original draft
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458 **Conflict of Interest**

459 The authors declare no conflict of interest.

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802 Table 1: Regression models for functional groups for (A): Detection probability estimation (p) and (B): Occupancy probability estimation (ψ). K is
803 number of parameters, AIC is the Akaike's Information Criterion, Δ AIC is the difference AIC from the best fit models, AIC.wt is model weight and
804 Cwt is cumulative model weights. Only the best detection model is shown. The best and worst occupancy models represent the extreme models that
805 contain the 90% confidence set of best-ranked ($Cwt \leq 0.90$). Because the number of occupancy models is too high, only the three best models and
806 the worst are shown based on the AIC value. The worst occupation model of each functional group was used in goodness of fit test (Supplementary
807 Material 4). The landscape variables TOA, PD, APS correspond to total occupied area, patch density (patches number per each 100 ha) and average
808 patch size (ha), respectively. The numbers that accompany the landscape variables indicate the land use type, where 1 = farmland; 3 = mixed forest;
809 4 = native forest; 6 = plantation forest; and 10 = isolated tree. The temporal variables MBS, T, RH, RPD and WV are minutes before the sunset
810 (min), temperature ($^{\circ}$ C), relative humidity (%), rainfall of the previous day (mm) and wind velocity (km/h), respectively. L is the type of landscape
811 (A or B).

Functional Trait	Models	AIC	Δ AIC	AIC.wt	Cwt
Granivores	A. Detection				
	$p(T:L+MBS+L) \sim \psi()$	3662.29			
	B. Occupancy				
	$p(T:L+MBS+L) \sim \psi(PD_3)$	3649.52	0	0.003	0.003
	$p(T:L+MBS+L) \sim \psi(PD_3+TOA_1)$	3650.25	0.73	0.002	0.005
	$p(T:L+MBS+L) \sim \psi(PD_3+APS+4:L)$	3650.35	0.84	0.002	0.007
	$p(T:L+MBS+L) \sim \psi(PD_3:L+PD_4:L+PD_6:L+APS_4:L+APS_{10}:L)$	3660.10	10.6	0	0.9
Insectivores	A. Detection				
	$p(L) \sim \psi()$	6903.50			
	B. Occupancy				
	$p(L) \sim \psi(PD_4+L)$	6887.67	0	0.005	0.005
	$p(L) \sim \psi(PD_4+PD_3:L+L)$	6887.79	0.12	0.005	0.01
	$p(L) \sim \psi(PD_4+PD_3:L+TOA_1+L)$	6887.95	0.28	0.005	0.015
	$p(L) \sim \psi(PD_4+PD_6:L+PD_{10}+APS_3+APS_4+APS_{10}+TOA_1:L+L)$	6898.71	11	0	0.9

	A. Detection				
	$p(\text{RPD:L+WV:L}) \sim \psi()$	2164.86			
	B. Occupancy				
Omnivores	$p(\text{RPD:L+WV:L}) \sim \psi(\text{PD}_3+\text{PD}_6:\text{L}+\text{PD}_{10}+\text{TOA}_1:\text{L}+\text{L})$	2131.66	0	0.01	0.01
	$p(\text{RPD:L+WV:L}) \sim \psi(\text{PD}_3+\text{PD}_6+\text{PD}_{10}+\text{TOA}_1:\text{L}+\text{L})$	2132.20	0.54	0.008	0.012
	$p(\text{RPD:L+WV:L}) \sim \psi(\text{PD}_3+\text{PD}_6:\text{L}+\text{PD}_{10}+\text{APS}_6+\text{TOA}_1:\text{L}+\text{L})$	2132.54	0.88	0.007	0.024
	$p(\text{RPD:L+WV:L}) \sim \psi(\text{PD}_3+\text{PD}_4:\text{L}+\text{PD}_6+\text{PD}_{10}:\text{L}+\text{APS}_3:\text{L}+\text{APS}_4+\text{APS}_6+\text{APS}_{10}:\text{L}+\text{TOA}_1+\text{L})$	2142.84	11.2	0	0.9
	A. Detection				
	$p(\text{RPD+WV}) \sim \psi()$	568.87			
	B. Occupancy				
Carnivores/Psicivores	$p(\text{RPD+WV}) \sim \psi(\text{PD}_3:\text{L}+\text{APS}_4:\text{L})$	561.28	0	0.004	0.004
	$p(\text{RPD+WV}) \sim \psi(\text{APS}_4:\text{L})$	561.29	0.01	0.004	0.009
	$p(\text{RPD+WV}) \sim \psi(\text{APS}_4:\text{L}+\text{APS}_{10})$	561.38	0.11	0.004	0.013
	$p(\text{RPD+WV}) \sim \psi(\text{PD}_3:\text{L}+\text{PD}_4+\text{PD}_6:\text{L}+\text{PD}_{10}:\text{L}+\text{APS}_4:\text{L}+\text{APS}_6+\text{APS}_{10}+\text{TOA}_1+\text{L})$	572.50	11.2	0	0.9
	A. Detection				
	$p(\text{T+WV+L}) \sim \psi()$	945.02			
	B. Occupancy				
Herbivores/Frugivores	$p(\text{T+WV+L}) \sim \psi(\text{APS}_{10}:\text{L}+\text{TOA}_1)$	938.10	0	0.002	0.002
	$p(\text{T+WV+L}) \sim \psi(\text{APS}_{10}:\text{L}+\text{TOA}_1:\text{L}+\text{L})$	938.42	0.32	0.002	0.004
	$p(\text{T+WV+L}) \sim \psi(\text{PD}_{10}:\text{L}+\text{TOA}_1+\text{L})$	938.43	0.33	0.002	0.006
	$p(\text{T+WV+L}) \sim \psi(\text{PD}_3+\text{PD}_6+\text{PD}_{10}:\text{L}+\text{APS}_4:\text{L}+\text{APS}_6+\text{APS}_{10}+\text{TOA}_1+\text{L})$	948.54	10.4	0	0.9

812

813

814 Figure Captions

815

816 Figure 1: Example pictures of the spatial heterogeneity in the study area of this project. (A) Large
817 heterogeneity area with relatively small crops, well-connected non-productive elements. (B)
818 Small heterogeneity area with relatively large crops and marginal non-productive elements,
819 dispersed and unconnected. Example raster map with buffer of 500 m radius to calculate the
820 landscape metrics with Fragstats.

821

822 Figure 2. Boxplots of observed and estimated bird richness associated to the two types of
823 landscape (A: heterogeneous; B: homogeneous) in La Araucania, Chile. The line inside the box
824 is the median and the whiskers is the maximum and minimum, respectively.

825

826 Figure 3. Relationships between the number of bird species (community size, N_{site}) and the
827 forest type in each landscape type in La Araucaria (Chile): a) native forest (%); b) mixed forest
828 (%); and c) plantations (%). Each point represents the richness of each count point surveyed (n
829 = 80), considering the two types of landscapes in different colour: A, heterogeneous (blue) and
830 B, homogeneous (green). Lines represent linear splines smooth.

831

832 Figure 4. Boxplot of estimated occupancy probabilities using observed landscape variables as
833 predictors for each functional trait group according to landscape types (A blue colour or B green
834 colour) resulting of model averaged parameter estimates (across 90% confidence set of best-
835 ranked regression models). Mann-Whitney test p -values are on top of the upper whiskers of the
836 boxes, which indicate statistically significant differences between medians. For
837 carnivores/piscivores and herbivores/frugivores medians comparison was not made, because
838 there was not equality of variances (Levene's test p -values equal to 2×10^{-3} and 4.7×10^{-5} ,
839 respectively).

840

841 Figure 5: The importance of each habitat variables in the occupancy probability of each
842 functional group for the two landscapes resulting of model averaged parameter estimates. Bars
843 indicate the 90% confidence intervals. If the bar does not cross the vertical dashed line, this
844 variable is statically significant. Estimates were calculated by averaging across 90% confidence
845 set of best-ranked regression models.

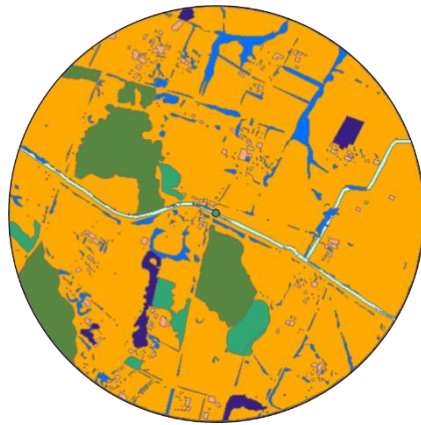
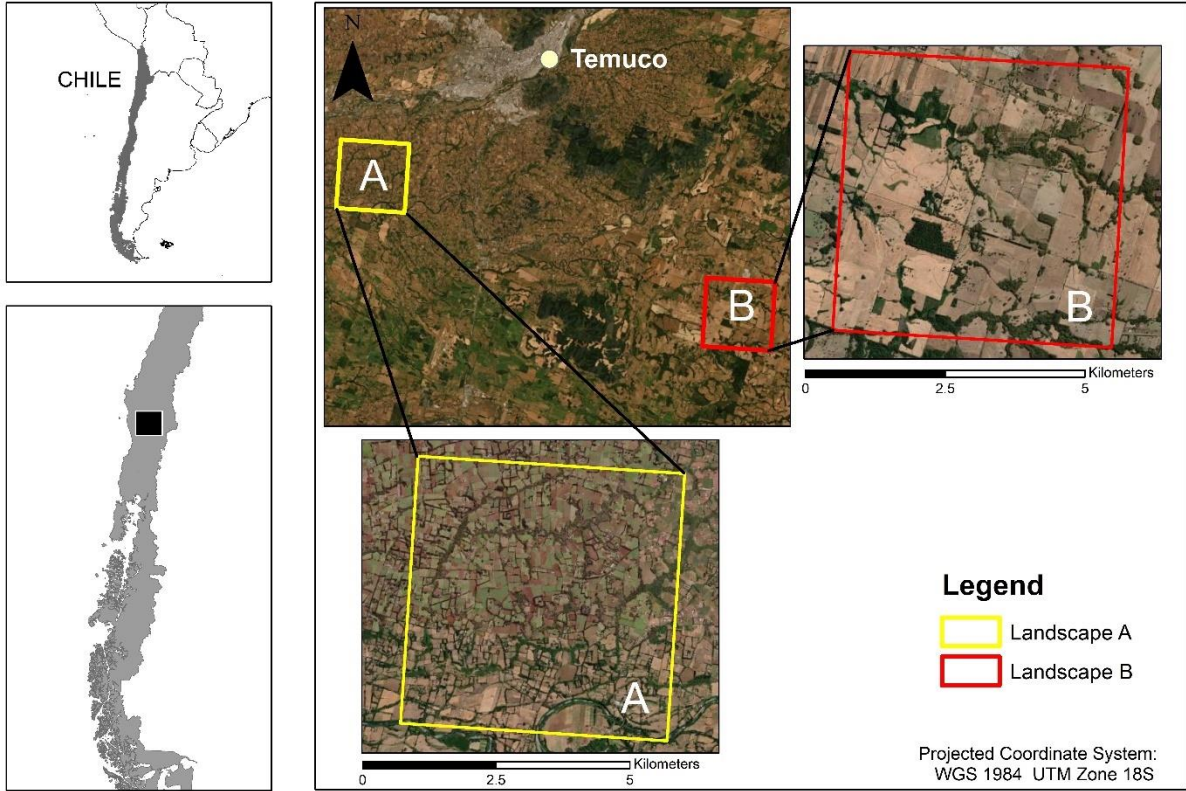
846

847 Figure 6: Panel plots for each functional traits showing of the occupancy probability in our study
848 area ($n = 80$) resulting of model-averaged predictions. It only shows the variables which are
849 significant for each landscape type according to the Figure 5. Lines show predicted covariate
850 effects, when all other variables do not influence. Estimates were calculated by averaging across
851 90% confidence set of best-ranked regression models. Shaded regions are unconditional 90%
852 confidence intervals.

853

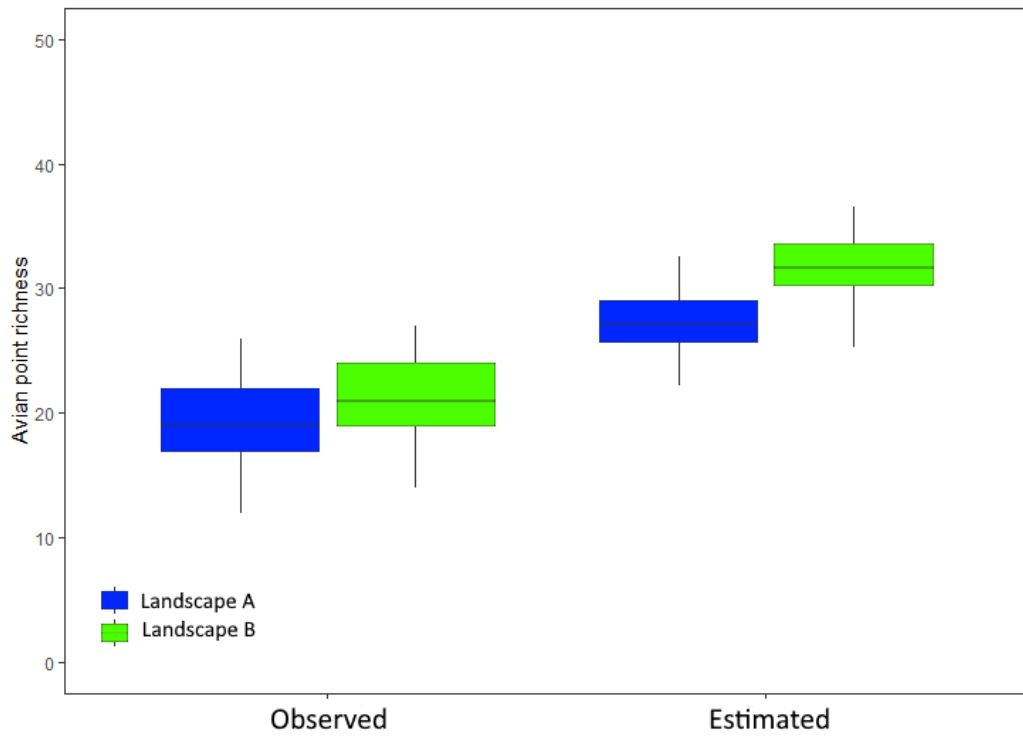
854 Figure 1

855



Raster map with buffer of 500 m radius

856 Figure 2

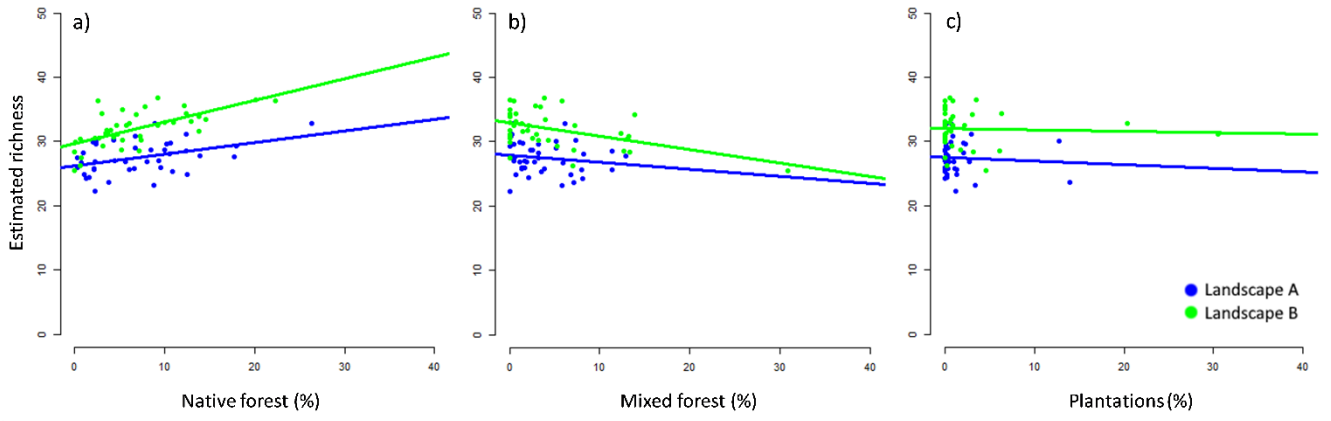


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860 Figure 3.

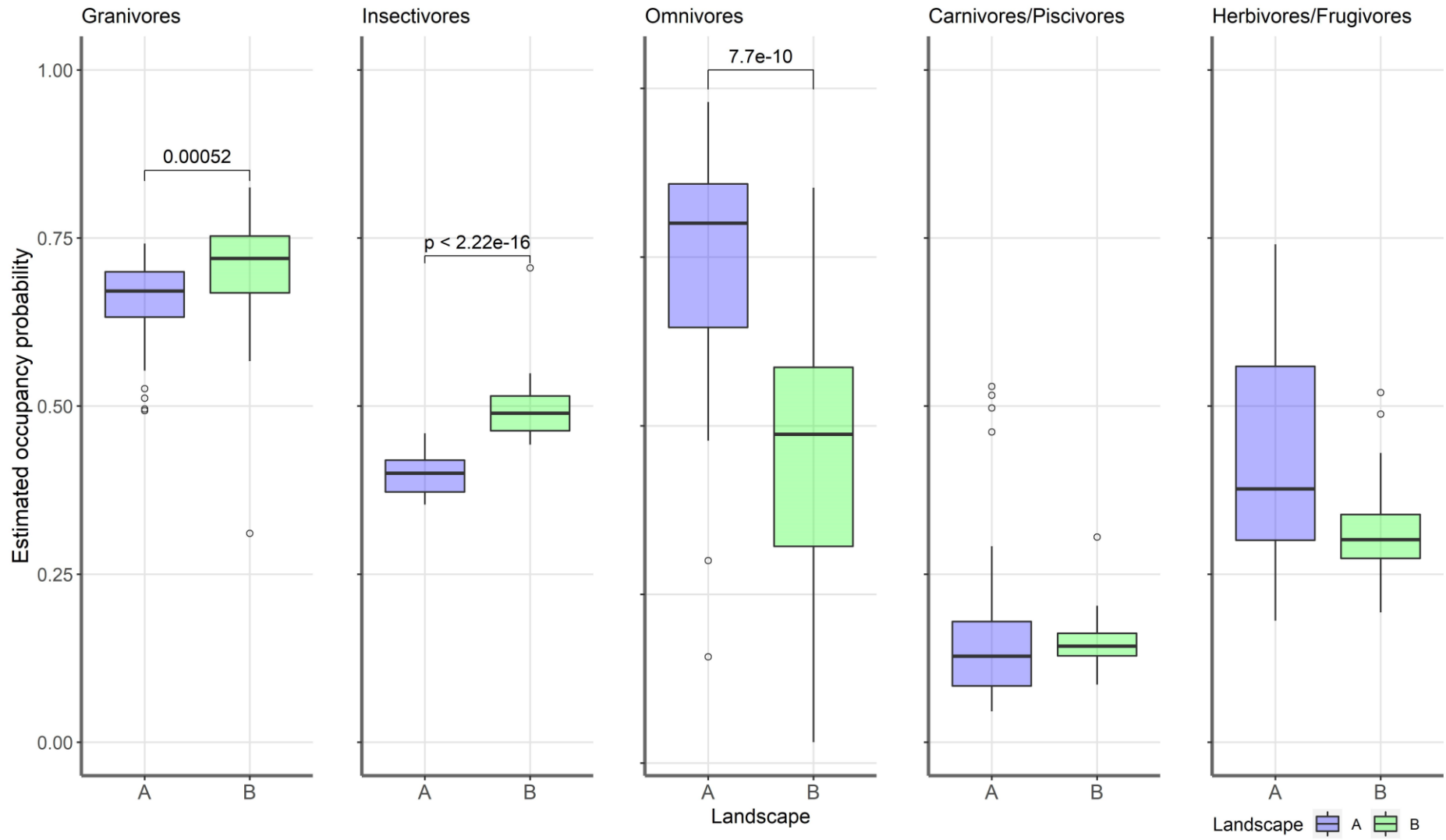


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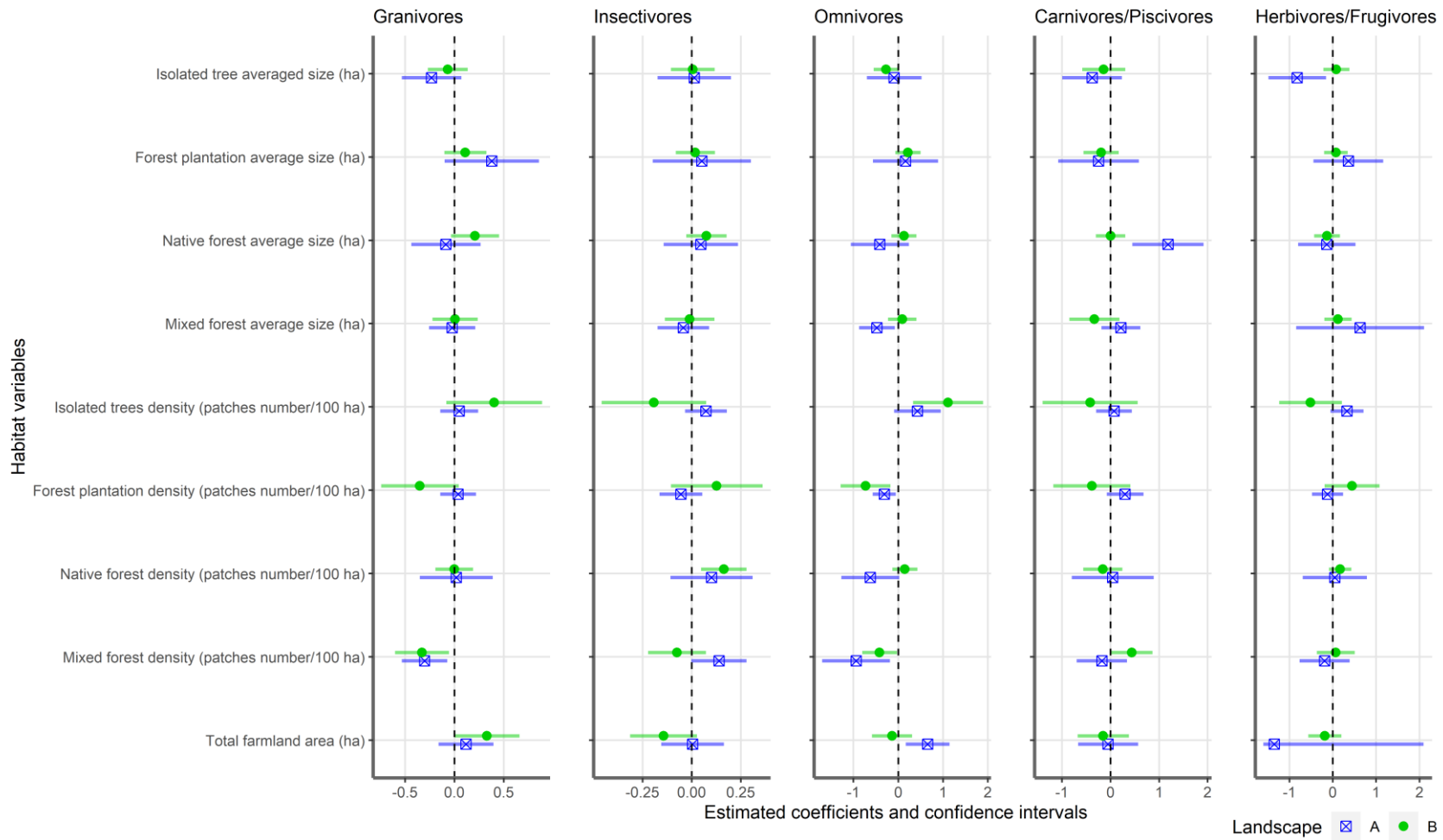
863

864 Figure 4:



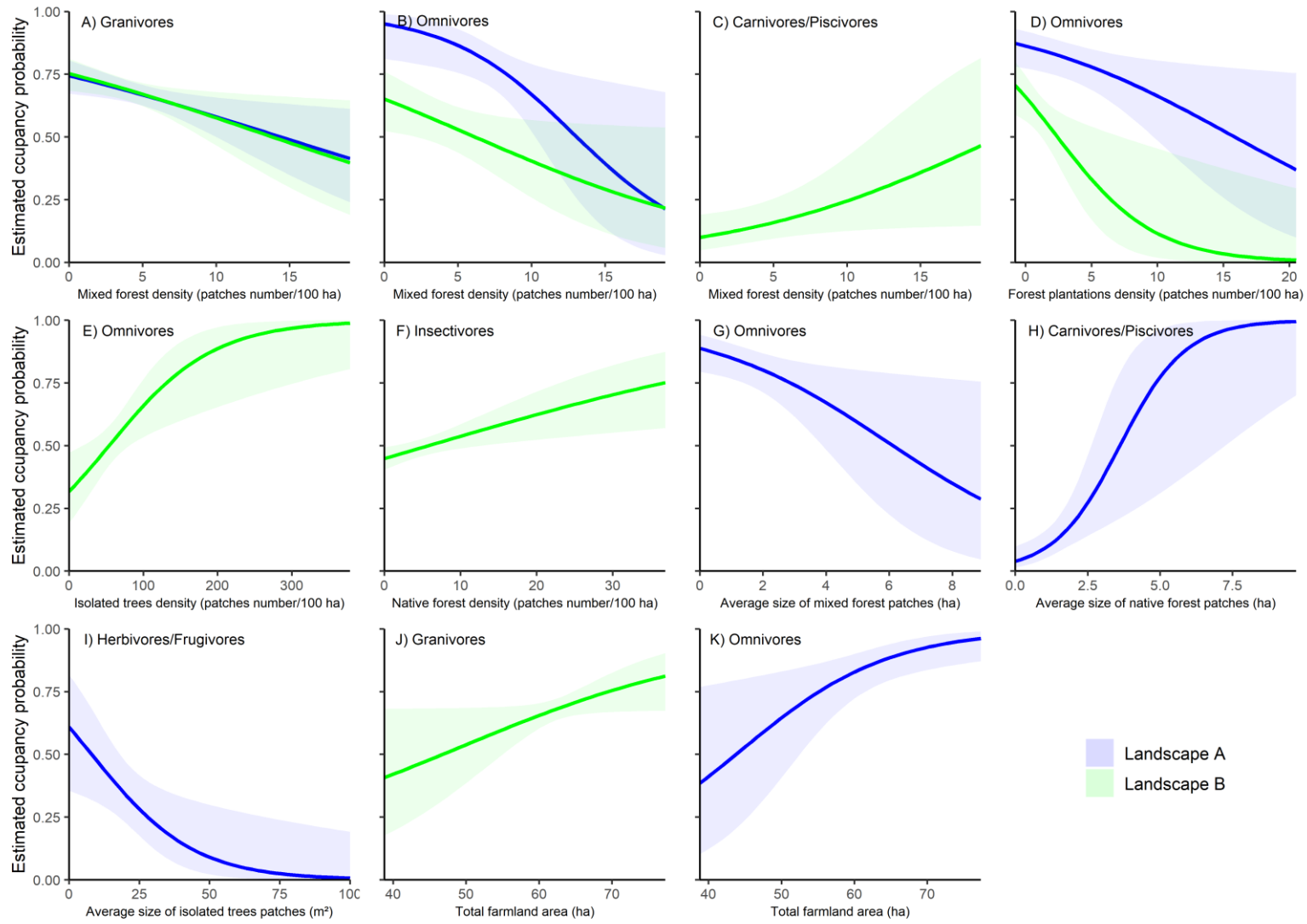
865

866 Figure 5:



867

868 Figure 6:



869

870 Supplementary Material 1: List of bird species recorded in the two landscape types and their functional traits, as well as observed occupancy and
871 estimated occupancy (Psi) and detection probability (p) for each bird species. Note that x is equal to the number of sample points in which a
872 species was detected. Naïve estimate is x divided by 80 (the number of plausible cells within which a species might occur).

873

Species	Code	Functional traits	Landscape A	Landscape B	X	Naïve Psi	Estimated Psi	Estimated p	Functional traits
<i>Agriornis lividus</i>	AGLIV	Insectivore		X	1	0.01	0.11	0.05	2
<i>Agelasticus thilius</i>	AGTHI	Insectivore	X	X	6	0.08	0.71	0.03	2
<i>Anas flavirostris</i>	ANFLA	Insectivore		X	1	0.01	0.15	0.04	2
<i>Anas georgica</i>	ANGEO	Herbivore/Frugivore	X	X	3	0.04	0.51	0.05	5
<i>Anairetes parulus</i>	ANPAR	Insectivore	X	X	65	0.81	0.92	0.41	2
<i>Anas sibilatrix</i>	ANSIB	Herbivore/Frugivore		X	1	0.01	0.20	0.07	5
<i>Aphrastura spinicauda</i>	APSPI	Insectivore	X	X	25	0.31	0.49	0.29	2
<i>Ardea alba</i>	ARALB	Carnivore/Piscivore	X	X	7	0.09	0.45	0.19	4
<i>Asio flammeus</i>	ASFLA	Carnivore/Piscivore		X	2	0.03	0.36	0.07	4
<i>Bubulcus ibis</i>	BUIBI	Insectivore	X	X	5	0.06	0.30	0.08	2
<i>Carduelis barbata</i>	CABAR	Granivore	X	X	36	0.45	0.47	0.18	1
<i>Callipepla californica</i>	CACAL	Granivore	X	X	37	0.46	0.28	0.23	1
<i>Cinclodes patagonicus</i>	CIPAT	Insectivore	X	X	22	0.28	0.48	0.24	2
<i>Cistothorus platensis</i>	CIPLA	Insectivore		X	1	0.01	0.02	0.77	2
<i>Columba araucana</i>	COARA	Herbivore/Frugivore	X	X	41	0.51	0.94	0.20	5
<i>Coragyps atratus</i>	COATR	Carnivore/Piscivore	X	X	15	0.19	0.95	0.06	4
<i>Colaptes pitius</i>	COPIT	Insectivore	X	X	40	0.50	0.91	0.17	2
<i>Curaeus curaeus</i>	CUCUR	Omnivore	X	X	65	0.81	0.67	0.39	3
<i>Diuca diuca</i>	DIDIU	Granivore	X	X	61	0.76	0.44	0.52	1
<i>Elaenia albiceps</i>	ELALB	Omnivore	X	X	68	0.85	0.76	0.40	3

<i>Elanus leucurus</i>	ELLEU	Carnivore/Piscivore	X	X	10	0.13	0.69	0.12	4
<i>Enicognathus leptorhynchus</i>	ENLEP	Granivore	X	X	37	0.46	0.18	0.29	1
<i>Falco femoralis</i>	FAFEM	Carnivore/Piscivore		X	4	0.05	0.56	0.06	4
<i>Falco sparverius</i>	FASPA	Insectivore	X	X	7	0.09	0.29	0.11	2
<i>Fulica armillata</i>	FUARM	Herbivore/Frugivore		X	1	0.01	0.05	0.78	5
<i>Geranoaetus polyosoma</i>	GEPOL	Carnivore/Piscivore	X	X	8	0.10	0.87	0.05	4
<i>Hymenops perspicillata</i>	HYPER	Insectivore	X	X	4	0.05	0.09	0.26	2
<i>Leptasthenura aegithaloides</i>	LEAEG	Insectivore	X	X	8	0.10	0.40	0.08	2
<i>Megaceryle torquata</i>	METOR	Carnivore/Piscivore		X	2	0.03	0.33	0.07	4
<i>Milvago chimango</i>	MICHI	Insectivore	X	X	67	0.84	0.94	0.43	2
<i>Mimus thenca</i>	MITHE	Omnivore	X	X	47	0.59	0.17	0.39	3
<i>Molothrus bonariensis</i>	MOBON	Omnivore	X	X	21	0.26	0.46	0.10	3
<i>Nothoprocta perdicaria</i>	NOPER	Omnivore	X	X	44	0.55	0.44	0.22	3
<i>Nycticorax nycticorax</i>	NYNYC	Carnivore/Piscivore	X		1	0.01	0.11	0.05	2
<i>Oxyura ferruginea</i>	OXFER	Carnivore/Piscivore		X	1	0.01	0.02	0.55	2
<i>Passer domesticus</i>	PADOM	Granivore	X	X	20	0.25	0.06	0.37	1
<i>Pardirallus sanguinolentus</i>	PASAN	Insectivore	X	X	9	0.11	0.83	0.04	2
<i>Parabuteo unicinctus</i>	PAUNI	Carnivore/Piscivore	X	X	6	0.08	0.82	0.04	4
<i>Phalacrocorax brasilianus</i>	PHBRA	Carnivore/Piscivore	X	X	2	0.03	0.40	0.06	4

<i>Phleocryptes melanops</i>	PHMEL	Insectivore		X	1	0.01	0.12	0.05	2
<i>Phytotoma rara</i>	PHRAR	Herbivore/Frugivore	X		24	0.30	0.86	0.16	5
<i>Plegadis chihi</i>	PLCHI	Insectivore		X	1	0.01	0.14	0.04	2
<i>Porphyriops melanops</i>	POMEL	Herbivore/Frugivore		X	2	0.03	0.17	0.17	5
<i>Podilymbus podiceps</i>	POPOD	Carnivore/Piscivore	X	X	2	0.03	0.14	0.32	4
<i>Pterotochos tarnii</i>	PTTAR	Insectivore		X	6	0.08	0.20	0.14	2
<i>Pygarrhichas albogularis</i>	PYALB	Insectivore		X	16	0.20	0.44	0.15	2
<i>Pygochelidon cyanoleuca</i>	PYCYA	Insectivore	X		2	0.03	0.30	0.04	2
<i>Rollandia rolland</i>	ROROL	Carnivore/Piscivore		X	1	0.01	0.09	0.31	4
<i>Scytalopus magellanicus</i>	SCMAG	Insectivore	X	X	24	0.30	0.47	0.29	2
<i>Scelorchilus rubecola</i>	SCRUB	Insectivore		X	21	0.26	0.35	0.63	2
<i>Sephanoides sephanoides</i>	SESEP	Herbivore/Frugivore	X	X	23	0.29	0.75	0.23	5
<i>Sicalis luteola</i>	SILUT	Granivore	X	X	76	0.95	0.90	0.52	1
<i>Sturnella loica</i>	STLOI	Omnivore	X	X	69	0.86	0.59	0.47	3
<i>Sylviorthorhynchus desmurri</i>	SYDES	Insectivore	X	X	16	0.20	0.40	0.21	2
<i>Tachycineta meyeni</i>	TAMEY	Insectivore	X	X	54	0.68	0.87	0.31	2
<i>Theristicus melanopis</i>	THMEL	Insectivore	X	X	75	0.94	0.97	0.62	2
<i>Troglodytes aedon</i>	TRAED	Insectivore	X	X	71	0.89	0.94	0.61	2
<i>Turdus falcklandii</i>	TUFAL	Omnivore	X	X	72	0.90	0.96	0.50	2
<i>Tyto alba</i>	TYALB	Carnivore/Piscivore		X	1	0.01	0.18	0.09	4
<i>Vanellus chilensis</i>	VACHI	Insectivore	X	X	78	0.98	0.98	0.86	2

<i>Veniliornis lignarius</i>	VELIG	Insectivore		X	1	0.01	0.11	0.05	2
<i>Xolmis pyrope</i>	XOPYR	Insectivore	X	X	28	0.35	0.92	0.11	2
<i>Zenaida auriculate</i>	ZEAUR	Granivore	X	X	75	0.94	0.93	0.47	1
<i>Zonotrichia capensis</i>	ZOCAP	Granivore	X	X	77	0.96	0.86	0.76	1
		Total	46	61					

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875 Supplementary Material 2: Descriptive statistics of landscape variables. Min, max, SD and CV are the minimum, maximum, standard deviation and
 876 coefficient of variation (%), respectively. The variables TOA, POA, PD, APS correspond to total occupied area, percentage of occupied area, patch
 877 density (patches number per each 100 ha) and average patch size (ha). The numbers that accompany the variables indicate the land use type, where
 878 1 = farmland; 2 = tree lines; 3 = mixed forest; 4 = native forest; 5 = wetland; 6 = plantation forest; 7 = river; 8 = road; 9 = urban; and 10 = isolated
 879 tree.

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Variable	Landscape A (n = 40)						Landscape B (n = 40)					
	Min	Max	Mean	Median	SD	CV %	Min	Max	Mean	Median	SD	CV %
TOA_1	47.7	72.7	62.5	62.0	6.2	10.0	38.8	77.3	63.1	64.0	7.5	11.9
TOA_2	1.4	6.3	3.5	3.2	1.0	29.7	0.2	6.5	2.7	2.6	1.6	58.5
TOA_3	0.0	10.1	3.2	2.4	2.8	88.0	0.0	24.2	3.5	1.8	4.9	139.6
TOA_4	0.2	20.7	5.8	5.3	4.5	77.5	0.0	17.5	5.3	4.2	4.2	79.1
TOA_5	0.0	0.3	0.0	0.0	0.1	632.5	0.0	2.5	0.1	0.0	0.5	377.1
TOA_6	0.0	11.0	1.1	0.3	2.3	210.6	0.0	24.0	1.8	0.3	4.5	258.1
TOA_7	0.0	6.1	0.5	0.0	1.3	247.8	0.0	4.7	0.4	0.0	1.1	283.6
TOA_8	0.0	2.5	0.9	0.8	0.4	48.0	0.0	1.4	0.9	0.8	0.2	27.4
TOA_9	0.2	2.9	0.8	0.6	0.6	70.1	0.0	1.1	0.3	0.3	0.3	86.4
TOA_10	0.0	1670	321.8	4.9	447.6	139.1	0.17	1558	254.6	0.6	522.3	205.2
POA_1	60.7	92.6	79.7	79.0	7.9	10.0	49.4	98.5	80.4	81.5	9.5	11.9
POA_2	1.8	8.1	4.4	4.1	1.3	29.7	0.2	8.3	3.4	3.3	2.0	58.5
POA_3	0.0	12.9	3.8	2.7	3.5	91.4	0.0	30.9	4.5	2.3	6.2	139.6
POA_4	0.3	26.3	7.1	6.7	5.7	80.3	0.0	22.3	6.8	5.3	5.3	79.1
POA_5	0.0	0.4	0.0	0.0	0.1	632.5	0.0	3.2	0.2	0.0	0.6	377.1
POA_6	0.0	14.0	1.4	0.3	2.9	208.9	0.0	30.5	2.2	0.4	5.8	258.1
POA_7	0.0	7.8	0.7	0.0	1.6	247.8	0.0	6.0	0.5	0.0	1.4	283.6
POA_8	0.0	3.1	1.2	1.0	0.5	45.6	0.5	1.8	1.1	1.1	0.2	21.6
POA_9	0.2	7.1	1.2	0.8	1.2	101.6	0.0	1.4	0.4	0.4	0.3	83.2
POA_10	0.0	0.2	0.1	0.1	0.1	59.3	0.0	0.2	0.1	0.1	0.0	48.5
PD_1	11.5	258.5	73.8	59.2	52.4	71.0	2.6	230.5	54.3	41.4	50.4	92.9
PD_2	42.0	300.6	111.1	86.0	65.7	59.2	10.2	175.8	57.1	43.3	38.8	68.0

PD_3	0.0	15.3	5.6	5.1	3.8	67.8	0.0	19.1	3.3	1.9	4.1	125.6
PD_4	1.3	15.3	4.9	5.1	3.0	60.1	0.0	36.9	5.2	3.8	5.9	113.5
PD_5	0.0	2.6	0.1	0.0	0.5	466.5	0.0	2.6	0.3	0.0	0.7	258.2
PD_6	0.0	14.0	3.1	1.3	3.2	105.8	0.0	6.4	1.2	1.3	1.5	123.8
PD_7	0.0	6.4	0.5	0.0	1.2	263.2	0.0	5.1	0.5	0.0	1.1	255.1
PD_8	0.0	20.4	2.6	1.3	3.9	148.2	1.3	3.8	1.5	1.3	0.6	38.0
PD_9	29.3	214.0	84.5	74.5	44.0	52.1	0.0	177.0	32.2	22.9	35.2	109.3
PD_10	0.0	378.3	87.5	47.1	101.9	116.5	8.9	179.6	57.8	47.1	41.4	71.6
APS_1	0.2	7.7	1.7	1.3	1.5	86.7	0.2	9.7	3.0	2.0	2.7	91.3
APS_2	0.0	0.2	0.1	0.0	0.0	62.5	0.0	0.2	0.1	0.1	0.0	50.5
APS_3	0.0	8.9	0.9	0.5	1.4	163.5	0.0	5.5	1.1	0.7	1.4	128.5
APS_4	0.2	3.8	1.5	1.4	0.9	62.1	0.0	9.7	1.7	1.3	1.7	98.6
APS_5	0.0	0.3	0.0	0.0	0.1	494.4	0.0	1.3	0.1	0.0	0.2	326.4
APS_6	0.0	2.5	0.4	0.2	0.6	144.7	0.0	6.0	0.9	0.2	1.6	186.5
APS_7	0.0	6.1	0.4	0.0	1.2	1.5	0.0	4.7	0.2	0.0	0.8	345.1
APS_8	0.0	1.6	0.7	0.8	0.4	0.1	0.2	1.4	0.8	0.8	0.2	30.0
APS_9	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	209.4
APS_10	0.0	48.0	16.4	13.0	11.1	122.5	6.0	100.0	22.3	16.5	17.5	78.6

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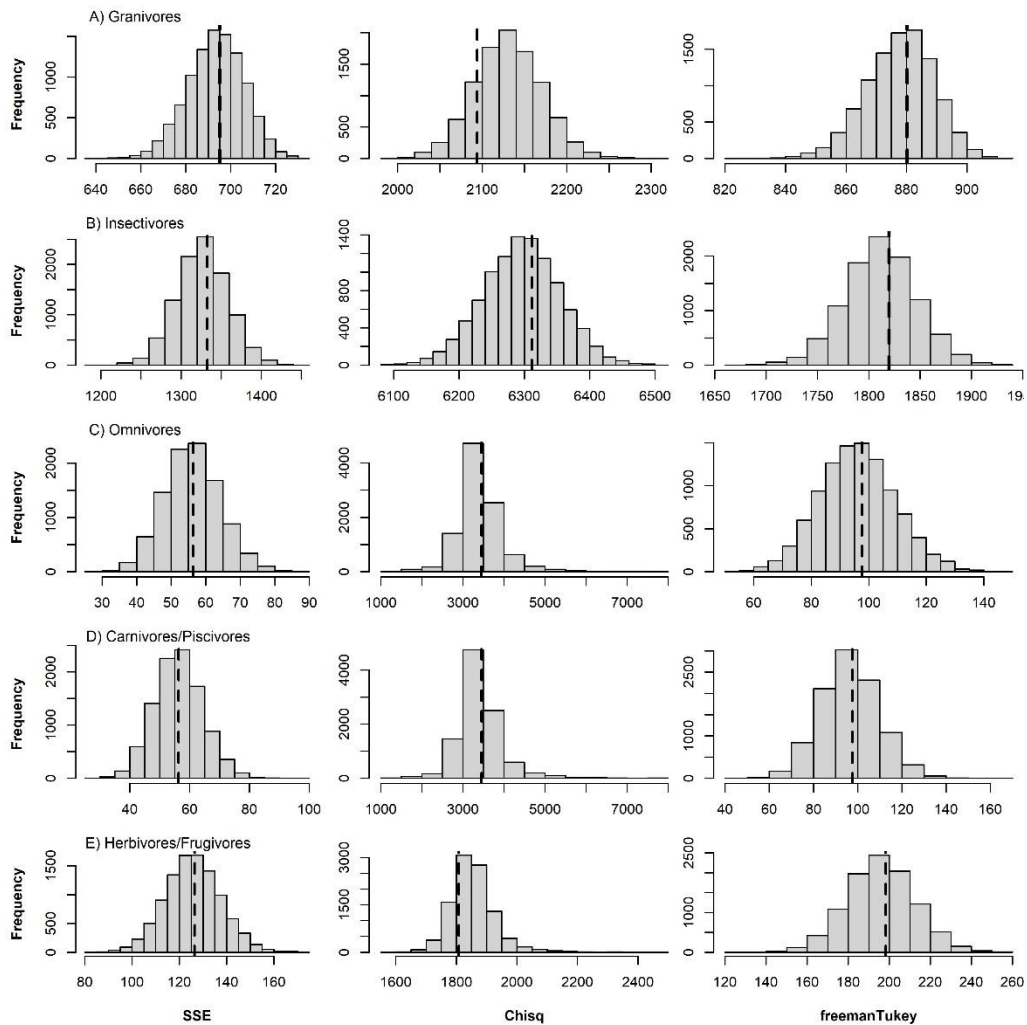
884 Supplementary material 3: Descriptive statistics of temporal variables (n = 400). Min, max, SD
885 and CV are the minimum, maximum, standard deviation and coefficient of variation (%),
886 respectively.

887

Variable	Min	Max	Mean	Median	SD	CV
Minutes before the sunset (min)	0.0	169.0	73.1	73.0	40.5	55.5
Temperature (°C)	7.5	28.4	18.0	17.7	4.2	23.3
Rainfall of the previous day (mm)	0.0	9.1	0.7	0.0	2.0	267.7
Wind velocity (km/h)	0.2	14.8	7.5	7.8	2.8	37.8

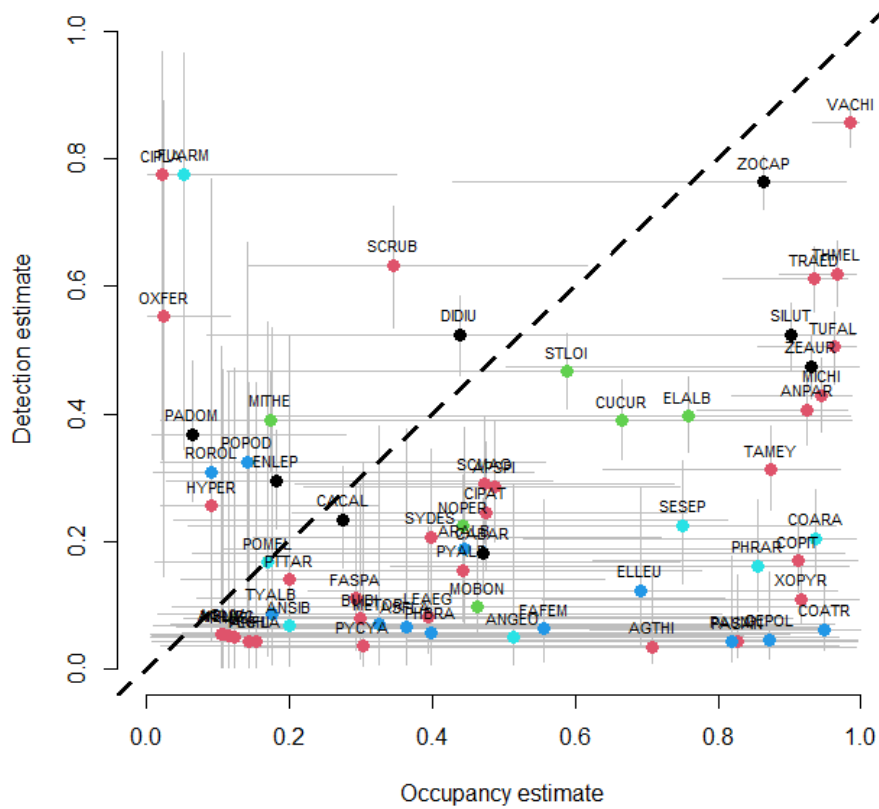
888

889 Supplementary Material 4: Graphical assessment of model fit by parametric bootstrapping for
 890 each functional traits. The dashed line is the observed statistic. The histogram approximates the
 891 expected sampling distribution based on 10,000 bootstrap samples for each fit assessment.



892

893 Supplementary Material 5: Mean of estimates of occupancy probability Psi and detection
 894 probability p for bird species in La Araucaria (Chile) considering the different functional traits
 895 in colours: omnivores (green); insectivores (red); granivores (black); carnivores/piscivores
 896 (blue); herbivores/frugivores (blue light).



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