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Foraging traces as an indicator to monitor wild boar impact on ground nesting birds.

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1 **Working paper- Foraging traces as an indicator to monitor wild boar**
2 **impact on ground nesting birds.**

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30

31 Conceived and designed the experiments: FR. Analyzed the data: FR and JMR. Contributed
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33

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48 **Foraging traces as an indicator to monitor wild boar impact on**
49 **ground nesting birds.**

50

51 **Abstract**

52 The successful management of large herbivores requires the monitoring of a set of indicators of
53 ecological change describing animal performance, relative animal abundance, and ungulate impact
54 on habitat. Wild boar populations increases have been spectacular in many countries including
55 France. Wild boars can have a substantial environmental impact on many ecosystem components
56 including birds, but indicators to monitor such impact are currently lacking. In this paper, we
57 examined the usefulness of monitoring the wild boar foraging traces made during their rooting
58 activity to evaluate their impact on ground nesting birds, and developed a simple indicator of
59 kilometric foraging (I_kF) traces found on transects. This study measured the effect of wild boar on
60 bird abundances, controlling for vegetation characteristics (shrub density, time since last clear-cut).
61 Using standardized spot-counts, we found that ground nesting bird abundances were negatively
62 correlated to I_kF. Our results showed a significant decrease of 44,7% in ground nesting bird
63 abundances in areas strongly foraged by wild boars. By contrast, abundances of birds depending of
64 ground or tree for food resource, but nesting out of reach of wild boars, were not correlated to
65 foraging traces. We thus conclude that I_kF is a simple and suitable indicator that can be used at
66 large scales to monitor wild boar impact on ground nesting birds.

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70 **Keywords**

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72 ungulates; birds; wild boar; ungulates management; over-abundance

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76 **List of abbreviations**

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78 AIC: Akaike information criterion

79 Cut: time since last clear-cut

80 GN: ground nesting

81 TN: tree nesting

82 GF: ground foraging

83 TF: tree foraging

84 IEC: indicator of ecological change

85 I_kF: indicator of kilometric foraging (continuous variable)

86 I_kF⁻: areas with low densities of foraging traces

87 I_kF⁺: areas with high densities of foraging traces

88 S: index of shrub density (continuous variable)

89 S⁻: areas with low shrub density

90 S⁺: areas with high shrub density

91 Tim: timing of spot-count (early vs. late spring)

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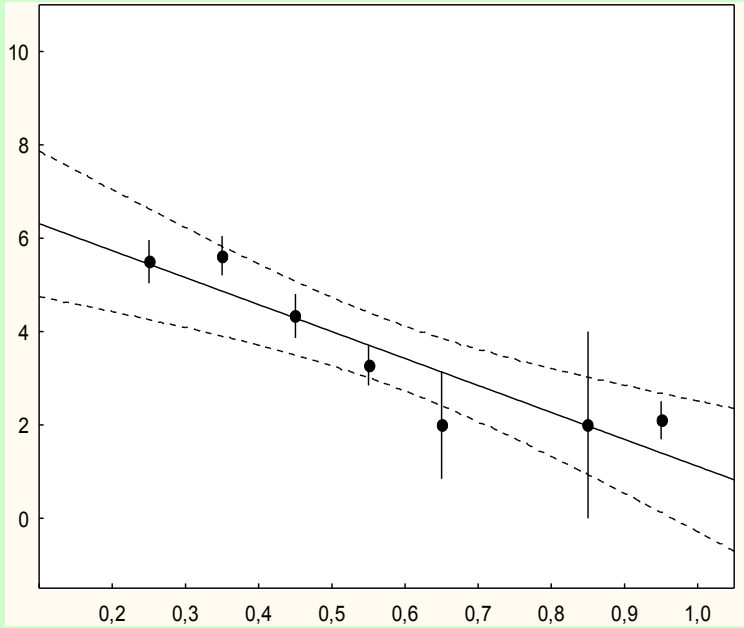
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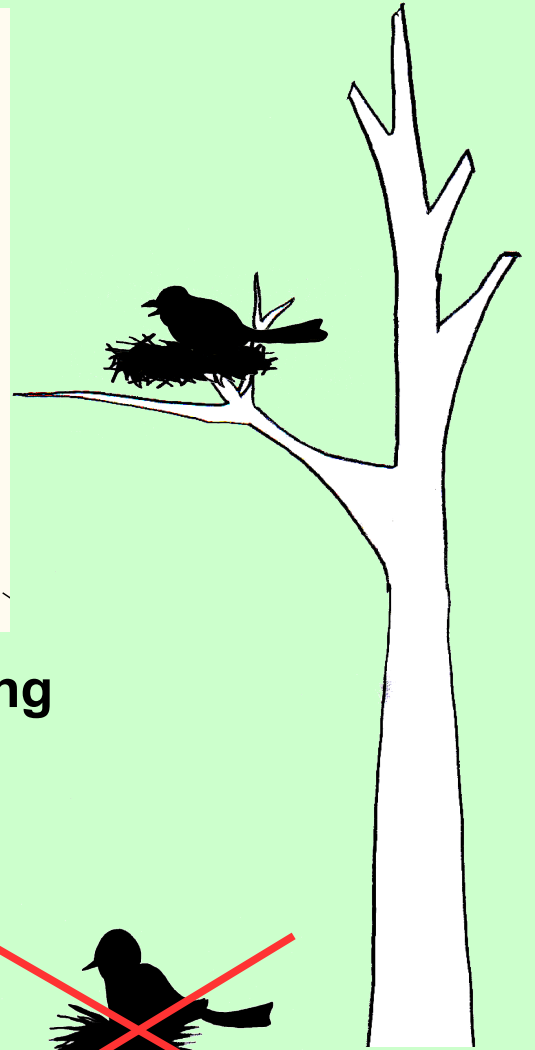
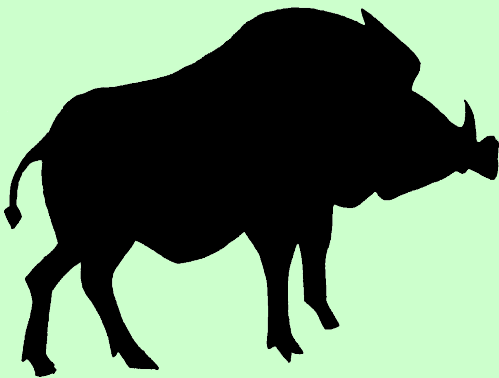
95 **Graphical abstract**

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Ground nesting birds



Indicator of wild boar foraging



147 1. Introduction

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149 In past decades, the decline of temperate bird species in Western Europe and in North America has
150 attracted much concern (Birdlife International, 2004; Julliard *et al.*, 2004a; Fuller *et al.*, 2007a;
151 Gregory & van Strien, 2010; EBCC, 2015). The causes of the observed decline of temperate
152 songbirds are complex. Habitat quality loss may be responsible of the observed decline of forest
153 (Fuller *et al.*, 2007a; Gregory *et al.*, 2007; Quine *et al.*, 2007; Fuller, 2012) and farmland songbirds
154 (Chamberlain & Fuller, 2000; Chamberlain *et al.*, 2000; Voříšek *et al.*, 2010). In addition, climate
155 change poses a threat to some bird species (Julliard *et al.*, 2004b; Jiguet *et al.*, 2007; Gregory *et al.*,
156 2009).

157

158 In parallel to the decline of temperate songbirds, wild ungulate populations increases have been
159 spectacular in many regions of North America (Côté *et al.*, 2004; Rawinski & Square, 2008) and
160 Western Europe (Milner *et al.*, 2006; Massei *et al.*, 2015). The main causes proposed for wild
161 ungulates over-abundance (as defined by Côté *et al.*, 2004) are climate change, the local
162 extermination of natural predator species, supplementary feeding, and changes in agricultural,
163 sylvicultural and game management practices (Flueck, 2000; Milner *et al.*, 2006; Servanty *et al.*,
164 2009; Massei *et al.*, 2015). For example in Europe, wild boar (*Sus scrofa*) populations reach
165 historical peaks (Saint-Andrieux *et al.*, 2012; Massei *et al.*, 2015). The impact of over-abundant
166 wild boar on conservation and economic interests include spread of diseases to livestock and
167 people, vehicle collisions, and damage to crops, as well as reduction in plant and animal abundance
168 and richness (Bourcet *et al.*, 2003; Massei & Genov, 2004; Brandt *et al.*, 2006; Vignon &
169 Barbareau, 2008; Barrios-Garcia & Ballari, 2012). As wild boar populations increase in many
170 European countries (Massei *et al.*, 2015), mitigating wild boar impact on environment and
171 economic interests will present a significant challenge. What are the known consequences of large
172 ungulates over-abundance on ecosystems? Numerous studies focus on high deer density impacts on
173 habitats. Deer over-abundance has strong effects on plant communities, vegetation structure, and on
174 abundance of many invertebrate taxa (Fuller & Gill, 2001; Gill & Beardall, 2001; Côté *et al.*, 2004;
175 Joys *et al.*, 2004; Allombert *et al.*, 2005a; Stockton *et al.*, 2005; Dolman & Wäber, 2008; Holt *et al.*,
176 2011; ONF, 2013). Such changes in the woodland ecosystem affecting food and safe nesting
177 availability are responsible for the negative cascading effects on songbird abundance and
178 biodiversity (McShea and Rappole, 2000; Allombert *et al.*, 2005b; Hewson *et al.*, 2011; Holt *et al.*,
179 2011; Martin *et al.*, 2011). As omnivorous mammals, wild boars can have a substantial
180 environmental impact and may affect many ecosystem components (Massei and Genov, 2004).
181 Their rooting activity can remove the herbaceous cover (Bratton, 1975; Howe *et al.*, 1981; Carpio
182 *et al.*, 2014a), and cause a decrease of tree regeneration (Gomez *et al.*, 2003). Wild boars negatively
183 affect rodents by direct predation on juveniles (Schley and Roper, 2003) or/and competition for
184 food resource (Focardi *et al.*, 2000; Carpio *et al.*, 2014a). Wild boar over-abundance has cascading
185 effects on threatened predators such as wildcat (*Felis silvestris*) by reducing rodent prey availability
186 (Lozano *et al.*, 2007). Previous studies on wild boar-birds interactions in their native range focused
187 mainly on game bird species (Massei & Genov, 2004). Wild boar predated capercaillie (*Tetrao*
188 *urogallus*) and hazel grouse (*Tetrastes bonasia*) nests (Saniga, 2002; Saniga, 2003) and simulated
189 nests of red-legged partridge (Carpio *et al.*, 2014b). As over-abundant wild boar preys on
190 everything on the ground, their negative effects both in their native and introduced range are thus
191 well established (Barrios-Garcia & Ballari, 2012).

192

193

194 To manage wild ungulates ecological footprint, it is necessary firstly to evaluate population size and
195 secondly to precisely measure the ungulates impact through environmental indices. Monitoring
196 changes in these indices provides a basis for setting management objectives (Morellet *et al.*, 2007).
197 Large herbivores populations are usually controlled through hunting in Europe (Milner *et al.*, 2006;

198 Massei *et al.*, 2015). Capture-Mark-Recapture methods (CMR) provide an efficient approach to
199 estimate reliably population size, but are extremely time-consuming and expensive when applied to
200 large herbivores (Buckland *et al.*, 2000). Distance sampling methods are less time-consuming but
201 suffer from less accuracy and precision (Buckland *et al.*, 2000; Thomas *et al.*, 2010). Both methods
202 do not provide any functional information on the population-habitat system, as population size *per*
203 *se* does not provide information on the environment. In addition, the primary concern for ungulates
204 managers is often to prevent over-abundance of herbivore populations (Myserud, 2006; Nichols &
205 Williams, 2006) rather than predicting accurately how many animals may be harvested or not.
206 Researchers have recently developed tools for managing ungulate populations, which are based on
207 the monitoring of a set of indicators of ecological change (“IEC”; Cederlund *et al.*, 1998; Morellet
208 *et al.*, 2007). An IEC is an easily measured parameter, sensitive to changes in the relationship
209 between the population and its resources and habitat (Cederlund *et al.*, 1998; Dale & Beyeler,
210 2001). Morellet *et al.* (2007) suggested tracking over time the variations of at least three categories
211 of IECs: one describing animal performance, another describing herbivore impact on environment,
212 and the last describing relative animal abundance. A set of IECs has been first developed for deer
213 management (Morellet *et al.*, 2007), but such a set of indicators is currently lacking for wild boar.
214 Recently, Acevedo *et al.* (2007) developed a method describing wild boar abundance based on the
215 frequency of fecal droppings. However, IECs describing wild boar impact on fauna, flora,
216 biodiversity and habitat are currently lacking. The main objective of this study was thus to find such
217 an IEC to monitor the impact of wild boar on avifauna.

218 We made the hypothesis that H1) wild boar preying activity should disturb ground nesting birds and
219 thus decrease their abundances in highly foraged areas. Tree nesting birds are out of reach of wild
220 boars, so we hypothesized that H2) tree nesting birds abundances should not negatively correlate
221 with wild boar foraging activity. As wild boars may have an impact on birds by food competition,
222 we investigated H3) the impact of wild boars on birds depending of ground for food resource, but
223 nesting out of reach of wild boars.

224

225 **2. Material and methods**

226 **2.1 Study area**

227

228 The forest massif of Morières / Siou Blanc is situated in southeastern France (8638-ha forest area;
229 altitude 500-650m, Figure S1), in an area with low human density, within which, there are only a
230 few isolated houses. The climate of Morières / Siou Blanc is Mediterranean, with hot and dry
231 summers, mild winters and moderately rainy autumns and springs (mean maximal temperature in
232 July = 27,6°C; mean minimal temperature in July = 19,4°C; maximal annual mean = 18,8°C;
233 minimal annual mean = 12,1°C; mean precipitation in July = 5,6 mm; annual mean 613,4 mm)
234 (Météo France). The forest massif exhibits Mediterranean vegetation, typical of southeastern
235 France. It displays various profiles of vegetation, according to forest management stages. It is a
236 coppice forest, with holm oak (*Quercus ilex*), downy oak (*Quercus pubescens*) and a large diversity
237 of shrubs. Some areas are dominated by various pines (*Pinus* spp.) and by xeric species of scrubs
238 (*Rosmarinus officinalis*, *etc.*). Numerous stands contained a mix of pines and oaks.

239

240 Sport hunting is an important activity in this forest massif, mainly focused on big game species,
241 especially wild boar. Wild boar abundance varies significantly throughout the massif, with high
242 numbers harvested (6,3 wild boars killed/km²/year). In contrast, roe deer (*Capreolus capreolus*) is
243 scarcer (0,9 animals killed/km²/year), and red deer (*Cervus elaphus*) is absent. A small cluster of
244 approximately 50 fallow deer (*Dama dama*), has been recently introduced by game managers.
245 Livestock animals are absent in the prospected area.

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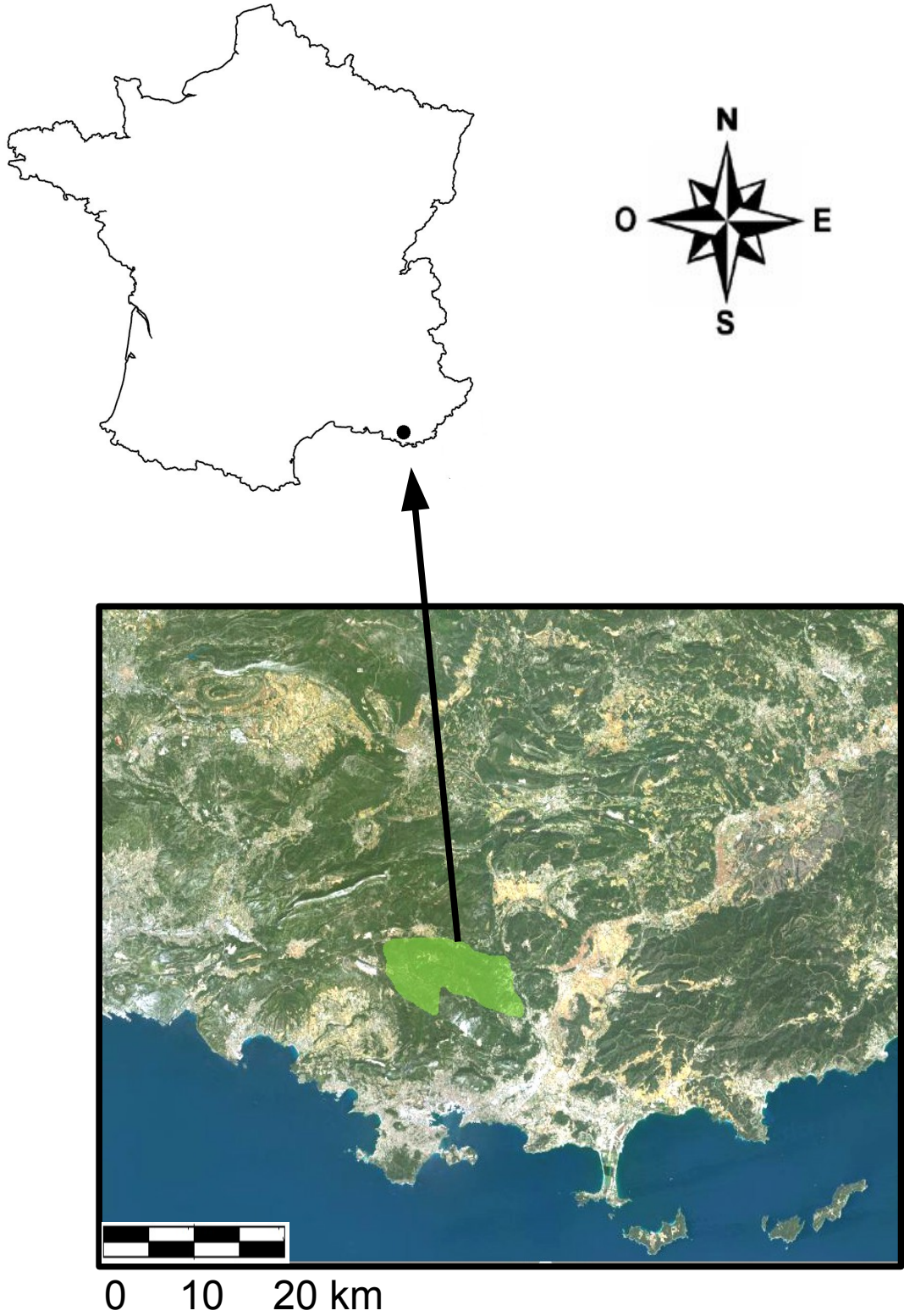


Figure S1: Location of the study area in southeastern France, showing layout of forest massif Of Siou Blanc les Morrières.

300 2.2 Sampling protocol

301

302 Twenty-six stations were randomly selected. They are evenly spread over the whole forest massif,
303 without prior knowledge of the vegetation, structure or bird communities. Ground visits were
304 organized during the winter season and the positions of the stations were recorded using GPS to
305 allow their subsequent localisation. The stations were in the main body of the forest of Morrières /
306 Siou Blanc and were therefore surrounded on all sides by woodlands. Around each station, a 1 km
307 survey transect was set-up to search for wild boar foraging traces. The survey routes were along
308 pre-existing paths or forest roads of 1-5m width (allowing the possibility for cars or people to enter
309 to the different places).

310

311 Each survey route was divided into 100m-length segments, yielding 10 segments per transect. In
312 each segment, the presence of wild boar foraging traces was recorded, which allowed the derivation
313 of a simple indicator of kilometric foraging (I_{kf}, a continuous variable), that is, the number of
314 segments with wild boar foraging traces/10 segments. This methodology gives an index derived on
315 the frequency of occurrence (see a similar methodology based on fecal droppings in Acevedo *et al.*,
316 2007). When signs of rooting activity were seen and occupied a contiguous ground area larger than
317 2m², the segments were considered as positive. Stations with I_{kf} ranging from 0 to 0,5 were
318 considered as low wild boar traces (I_{kf}-). Stations with I_{kf} from 0,6 to 1 were considered as high
319 wild boar traces (I_{kf}+). These two level factor were used for GLM analysis (see below, statistics
320 section). Two measures of I_{kf} were made during spring for each station, between 1st April and 30th
321 May 2012. We measured I_{kf} the same days as bird surveys (see 2.3). We found 25 counts with low
322 densities of wild boar foraging traces (I_{kf}-), and 27 counts with high wild boar foraging traces
323 (I_{kf}+).

324

325 Along each survey route, we measured variables related to habitat structure, *viz.*, tree cover, density
326 of shrub cover, average tree and shrub heights. Based on results of previous bird studies (see below
327 "*Vegetation data*" section), we expected these variables to play an important role.

328

329 2.3 Bird survey

330

331 Spot-counts were all done by the same observer (FR). Counts were carried out in each plot during
332 two visits in spring 2012 ; point-count times were separated by an interval of at least four weeks to
333 detect both early and late breeding birds (1st-15th April and 15th-30th May), following the
334 recommendations of the French Breeding Bird Survey of the National Museum of Natural History
335 (MNHN, 2012; Jiguet *et al.*, 2012). Counts were performed within 1-4h after sunrise. Point-count
336 stations (n = 26) were positioned a minimum of 500m apart to avoid overlap of bird counts. To
337 minimize a potential edge effect (Ries and Sisk, 2004), the center of all study plots were situated at
338 least 100m from the edge of forest alleys. We used 100m fixed-radius point-counts that lasted 5
339 minutes to allow for a good estimation of relative bird abundances, following the methodology
340 detailed in Blondel *et al.*, 1970 and Jiguet *et al.*, 2012. Repeating point-counts throughout the
341 breeding season improved estimation of species richness and relative abundances of birds (Dettmers
342 *et al.*, 1999; Drapeau *et al.*, 1999; Bonthoux and Balent, 2012). Points were recorded in the reverse
343 order during the second count to avoid introducing temporal bias between different stations. Since
344 all of the plots were located in core forest habitats with low visibility, birds were mostly detected
345 through their song. Individuals could be distinguished when several birds continuously sang at
346 different locations in the plot.

347

348 We excluded observations of raptors, *i.e.*, Golden Eagle (*Aquila chrysaetos*), Bonelli's Eagle
349 (*Aquila fasciata*), Short-toed Snake Eagle (*Circaetus gallicus*), Peregrine Falcon (*Falco*
350 *peregrinus*), Common Kestrel (*Falco tinnunculus*), Eurasian Sparrowhawk (*Accipiter nisus*),
351 corvids (*Corvus corrax* and *Corvus corone*), Common Cuckoo (*Cuculus canorus*), and Eurasian

352 Woodcock (*Scolopax rusticola*) as these species were poorly surveyed by the method used and/or
353 have life cycles that make them irrelevant to the objective of the study. Game birds (*Alectoris rufa*,
354 and *Phasianus colchicus*) were not recorded since their presence in a particular station may have
355 been due to game releases in the area.

356
357 To test the hypothesis that the wild boar impact on birds may result of direct predation, we
358 established a classification system that grouped the bird species encountered during the study into
359 two guilds according to the nesting habitat (*i.e.*, ground nesting *vs.* tree/shrub nesting birds). To test
360 the hypothesis that wild boar may impact birds through food competition, birds were grouped into
361 two guilds according to ground dependence for foraging and gleaning (*i.e.*, ground foraging *vs.* tree
362 foraging birds). Ground nesting birds were separated from ground foraging and tree foraging guilds
363 to avoid potential overlap of effects (*i.e.*, predation and food competition). Location of the nest-site
364 were taken from literature (Gregory *et al.*, 2007; Flitti *et al.*, 2009) and our own observations (See
365 Table S1). Four indicators were derived: an indicator for ground nesting birds (GN, n=201 birds),
366 and an indicator for tree nesting birds (TN, n=339 birds), an indicator for ground foraging birds
367 (GF, n=234 birds) and an indicator for tree foraging birds (TF, n=105 birds).

368
369 At the bird community level, we used species diversity as bird community indicator. Species
370 diversity was calculated using the Shannon index. The Shannon index reflects the number of
371 different species in a community and simultaneously takes into account how evenly the individuals
372 are distributed among those species (Lyashevskaya & Farnsworth, 2012).

373

374 2.4 Vegetation data

375

376 This study measured the effect of wild boar on birds, controlling for vegetation characteristics
377 (shrub density, time since last clear-cut). Vegetation data were collected during spring 2012, after
378 each bird count session. Measurements were taken at three sampling areas at each point-count
379 station, thus resulting in 78 sampling points. One was located at the point-count station itself, the
380 other two at random distances from the station (<50m). Mean values were calculated for each
381 variables listed. Shrub layer density of vegetation layer (0-1m) was estimated using a pole
382 graduated into 10cm sections and using a protocol adapted from Holt *et al.* (2011). The number of
383 sections obscured by vegetation was recorded when viewed from a distance of 10m. An index of
384 shrub density (S, a continuous variable) was derived using the number of segments obscured/10
385 segments, from 0 to 1. We used these scores to define two classes of shrub layer density: values
386 equal or smaller than 0,5 were considered as "low density" (S-), those equal or greater than 0,6 as
387 "high density" (S+); this two level factor was used for GLM analysis (see below, statistics section).
388 As avian community structure is largely determined by the growth stage and associated structural
389 characteristics of vegetation (Fuller *et al.*, 1989), vegetation age was controlled in each plot, using
390 local forest management archives. Time since last clear-cut (Cut) was used as a continuous variable
391 for statistical purpose (see below, statistics section).

392

393 As cervids are known to have an impact on birds (see Allombert *et al.*, 2005b; Holt *et al.*, 2011;
394 Martin *et al.*, 2011), signs of browsing were carefully searched in each sampling area. We
395 considered only shrub species or woody plants because we could not, with herbaceous species,
396 distinguish deer browsing from that of lagomorphs present in the study area. We recorded the
397 browsing pressure by assessing the number of eaten twigs versus all available twigs at a height
398 <1.80 m, following a methodology similar to those described elsewhere (Picard, 1988; Chevrier *et*
399 *al.*, 2012). A "browsing sign" is one bite by the animal on the tree, being evident as a fresh cut on
400 the twigs. As the browsing pressure was <1% in all the sampled plots, the cervids impact on the
401 avifauna was considered negligible in this study. All the assessments were done by the same person
402 (FR).

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Table S1
Classification of species in relation to foraging and nesting dependence

| Species acronym | Species name | Ground or Tree/shrub dependence | | Migration strategy |
|------------------------------------|---|---------------------------------|------------|--------------------|
| | | Nesting | Foraging | |
| <i>(a) Species</i> | | | | |
| <i>Cya cae (n=55)</i> | Eurasian Blue Tit (<i>Cyanistes caeruleus</i>) | Tree/shrub | Tree/shrub | Partial migrant |
| <i>Eri rub (n=69)</i> | Robin (<i>Erithacus rubecula</i>) | Ground | Ground | Partial migrant |
| <i>Fri coe (n=111)</i> | Common Chaffinch (<i>Fringilla coelebs</i>) | Tree/shrub | Ground | Partial migrant |
| <i>Lus meg (n=51)</i> | Nightingale (<i>Luscinia megarhynchos</i>) | Ground | Ground | Long migrant |
| <i>Par maj (n=50)</i> | Great Tit (<i>Parus major</i>) | Tree/shrub | Tree/shrub | Partial migrant |
| <i>Phy bon (n=81)</i> | Western Bonelli's Warbler (<i>Phylloscopus bonelli</i>) | Ground | Tree/shrub | Long migrant |
| <i>Syl atr (n=43)</i> | Eurasian Blackcap (<i>Sylvia atricapilla</i>) | Tree/shrub | Ground | Partial migrant |
| <i>Syl mel (n=26)</i> | Sardinian Warbler (<i>Sylvia melanocephala</i>) | Tree/shrub | Ground | Partial migrant |
| <i>Tur mer (n=34)</i> | Blackbird (<i>Turdus merula</i>) | Tree/shrub | Ground | Partial migrant |
| <i>(b) Guilds</i> | | | | |
| <i>Ground Nesting (GN, n=201)</i> | Common Nightingale; Robin; Western Bonelli's warbler | | | |
| <i>Tree Nesting (TN, n=339)</i> | Blackbird; Common Chaffinch; Great Tit; Eurasian Blackcap; Eurasian Blue Tit; Sardinian warbler | | | |
| <i>Ground Foraging (GF, n=234)</i> | Blackbird; Common Chaffinch; Eurasian Blackcap; Sardinian warbler | | | |
| <i>Tree Foraging (TF, n=105)</i> | Great Tit; Eurasian Blue Tit | | | |

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421 2.5 Statistics

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423 We tested for the effects of 1) wild boar foraging (Ikf), 2) shrub cover (S), 3) timing of spot-counts
424 (Tim) and 4) time since last clear-cut (Cut) (the factors of variation) on birds abundances, and
425 species diversity (the dependent variables) using general linear models (GLM). For the count data
426 (bird abundances), each model was tested assuming normal (Gaussian) and Poisson error. We
427 selected the best model using the Akaike Information Criterion (AIC). However, when the AIC of
428 two competing models differed by less than 2, we examined the residual sum of squares and
429 retained the simplest model in line with the rules of parsimony. We checked for the
430 homoscedasticity and normality assumptions of residuals. Raw data were not transformed, and real
431 birds counts were used throughout the paper. Timing of spot-count was entered as a two-level factor
432 (early vs. late) in all the models. Wild boar foraging and shrub cover were entered as a two-level
433 factors (Ikf- vs. Ikf+; and S- vs. S+, respectively) to investigate potential interactions between
434 factors. Entering wild boar foraging and shrub cover as continuous covariates led to the same
435 conclusions. Time since last clear-cut was entered as a continuous covariate (Cut) in all models.
436 Birds were grouped in functional guilds (See Table S1). We fitted several models including
437 interactive effects of Ikf, S, Tim and Cut. As a control of potential bias in our study, we tested the
438 effects of 1) S and 2) Tim (the factors of variation) on Ikf (the dependent variable) using GLM.
439 Statistical evaluation was carried out with the “Statistica” software using the functions “GLM”
440 (normal error) or “GLZ” (poisson error). Best results were obtained assuming normal error.
441 Nine bird species that were at least observed in one third of point-counts, were included in
442 statistical analysis. All results are expressed as mean \pm SEM. Differences were considered
443 significant at $P < .05$.

444

445

446 3. Results

447

448 3.1 General results

449 Wild boar foraging traces were observed in all surveyed transects. Maximum value of wild boar
450 indicator of kilometric foraging (Ikf) was of 1 with a high mean value of $0,60 \pm 0,03$, indicating a
451 relatively strong pressure of wild boar foraging in the study area considered as a whole. Twenty-five
452 spot-counts showed few wild boar traces (Ikf-), and 27 showed many traces (Ikf+, see methods
453 section). GLM analysis showed no relationships between Ikf and timing of spot-counts (Tim) or
454 shrub cover (S) (Adjusted $R^2=0,02$; $F=1,31$; $p=0,28$). We analyzed the abundances of 9 bird species
455 and 520 individuals (Table S1). Counts led to an average of $3,9 \pm 0,3$ SE ground nesting birds
456 (range 0-8); $6,5 \pm 0,3$ SE tree nesting birds (range 3-13); $4,5 \pm 0,3$ SE ground foraging birds (range
457 1-9); and $2,0 \pm 0,2$ SE tree foraging birds per spot (range 0-7).

458

459 3.2 Wild boar impact on birds

460 We tested the effects of Ikf, Tim, S and Cut on the abundance of ground nesting birds. Abundances
461 of TN, GF and TF birds were also tested as supplementary variables. The best model included the
462 variables Ikf, S and Tim; Cut, TN, GF and TF were rejected as explanatory variables (Table 1). The
463 observed changes were well explained by the GLM and were very statistically significant
464 (Adjusted $R^2=0,70$; $F=17,89$; $p<10^{-6}$). We found that the abundance values of GN birds were
465 negatively correlated with Ikf (Figure 1a). Entering Ikf as a two-level factor showed that there was
466 a significant decrease of 44.7% in ground nesting birds abundances in areas strongly foraged by
467 wild boars (Figure S2). GLM revealed the role of Ikf as being the main factor driving bird
468 abundance changes, and Tim and S as being secondary factors, in this order (Table 2). Test of
469 categorical variables interactions as explanatory variables were not significant (Table 2). The effect
470 of Tim showed that birds were more numerous in late spring. We observed that the negative effect
471 of wild boars on birds abundances was slightly greater in early spot-counts than late, but this result

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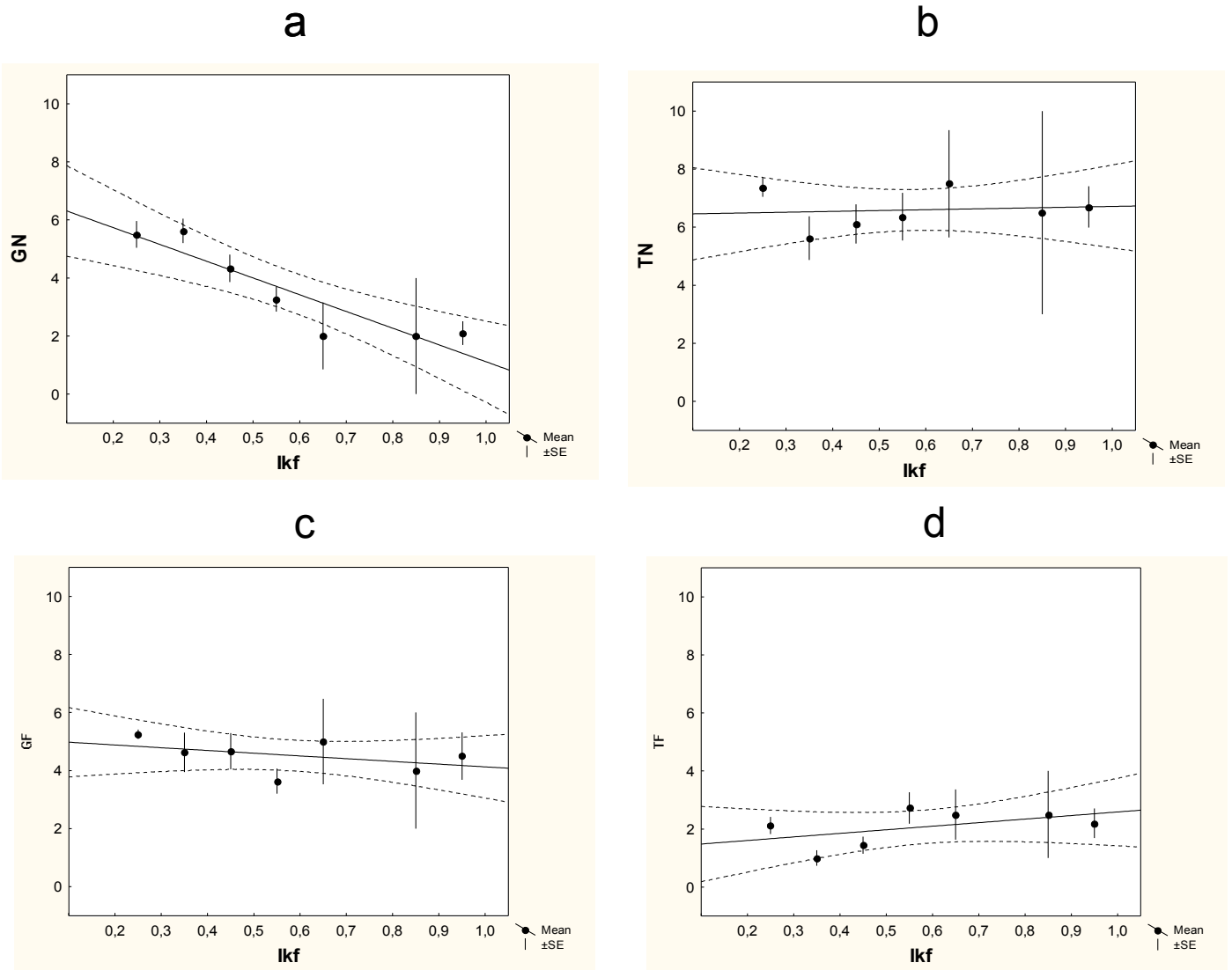
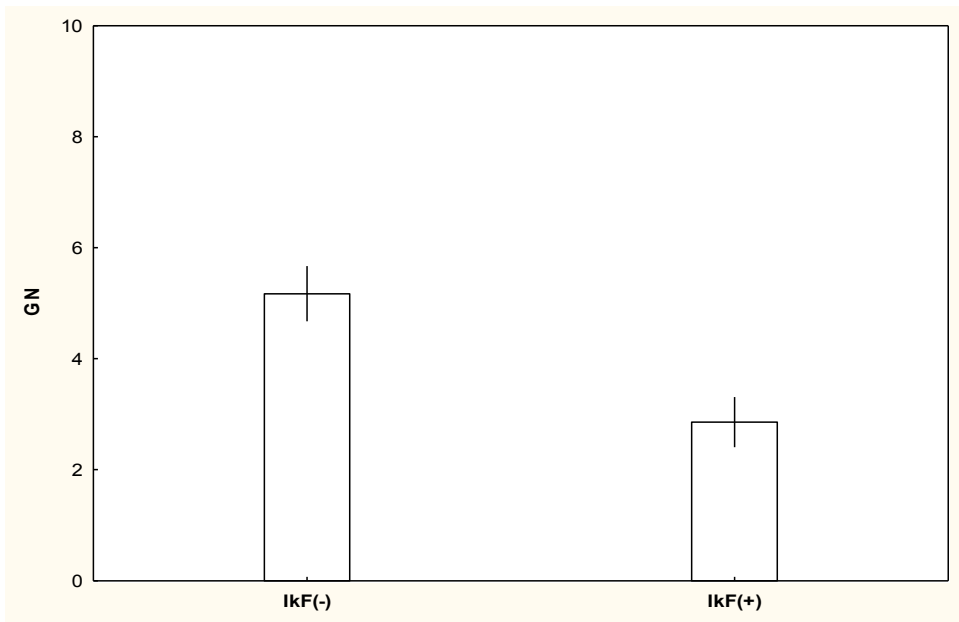


Figure 1: Variations of forest bird mean abundances in relation to wild boar foraging (Ikf). Predicted trends are presented as black lines with 95% confidence intervals (dashed lines). Bars indicate the standard error of mean. (a) ground nesting (GN) birds (b) tree nesting (TN) birds (c) ground foraging (GF) birds (d) tree foraging (TF) birds.

| Guild | selected variables | | | | | | | AIC | SS Residual |
|----------------------------|--------------------|----------|----------|-----|----|----|----|--------------|----------------|
| | lkf | Tim | S | Cut | TN | GF | TF | | |
| Ground Nesting (GN) | x | x | x | x | x | x | x | 161.7 | 52.1 |
| | x | x | x | | | | | 164.1 | 52.2 |
| | x | x | x | x | | | | 165.1 | 51.8 |
| | x | x | | | | | | 183.4 | 81.5 |
| | x | | x | | | | | 187.9 | 83 |
| | x | | | x | | | | 196.8 | 109.7 |
| | x | | | | | | | 197.7 | 115.3 |
| | | x | | | | | | 208.7 | 161.7 |
| | | | x | | | | | 214.3 | 167.4 |
| | | | | | | x | | 221.6 | 197.9 |
| | | | | | x | | | 221.9 | 199.7 |
| | | | | | | | x | 222.1 | 200.4 |
| | | | | x | | | | 222 | 200.4 |

526 **Table 1:** Candidate models explaining ground nesting birds abundance, with the Akaike
527 information criterion values (AIC) and the selected variables : wild boar foraging (lkf), timing of
528 spot-count (Tim), shrub density (S), time since last clear-cut (Cut), tree nesting birds abundance
529 (TN), ground foraging birds abundance (GF), tree foraging birds abundances (TF). The interactions
530 between wild boar foraging and timing (lkF*Tim), wild boar foraging and shrub density (lkF*S),
531 wild boar foraging, timing and shrub density (lkF*Tim*S) were also tested, but gave higher scores
532 of AIC(data not shown). Bold characters indicate the selected model.



554 **Figure S2:** Mean abundances of ground nesting (GN) in relation to wild boar foraging. lkf(-) areas
555 moderately foraged by wild boars; lkf(+) areas strongly foraged by wild boars.
556 Lines indicate standard error of mean. Results are highly significant ($p < 10^{-6}$); see Table 2 and
557 statistical analysis.

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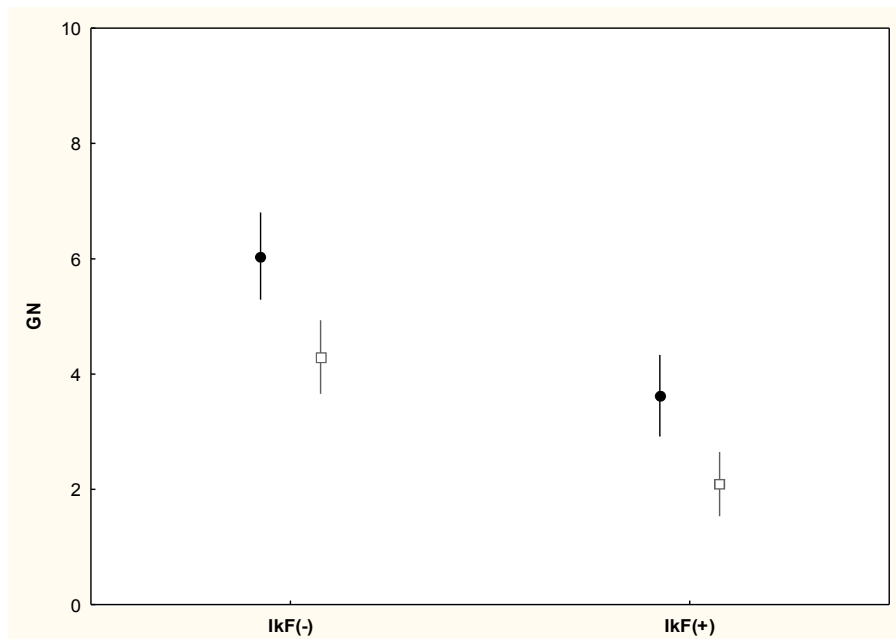
Estimates (sigma-restricted parameterization)

| Effect | Level of effect | Column | GN param. | ± SE | GN t | GN p | GN β | ± SE |
|------------------|-----------------|----------|-------------|------------|-------------|-----------------|--------------|-------------|
| <i>Intercept</i> | | 1 | 4,0 | 0,2 | 24,2 | 0,000000 | | |
| IkF | (-) | 2 | 1,2 | 0,2 | 7,0 | 0,000000 | 0,59 | 0,08 |
| Tim | early | 3 | -0,8 | 0,2 | -4,8 | 0,000018 | -0,41 | 0,08 |
| S | (+) | 4 | 0,8 | 0,2 | 4,9 | 0,000012 | 0,41 | 0,08 |
| IkF*Tim | 1 | 5 | 0,1 | 0,2 | 0,6 | 0,581136 | 0,05 | 0,08 |
| IkF*S | 1 | 6 | 0,1 | 0,2 | 0,3 | 0,744490 | 0,03 | 0,08 |
| Tim*S | 1 | 7 | 0,1 | 0,2 | 0,3 | 0,753062 | 0,03 | 0,08 |
| IkF*Tim*S | 1 | 8 | 0,0 | 0,2 | 0,2 | 0,831509 | 0,02 | 0,08 |

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Table 2 : β coefficients and statistical parameters of the most parsimonious GLM model to explain ground nesting (GN) birds abundance using wild boar foraging traces (Ikf), Timing (Tim) and shrub density (S) as predictors. Bold characters indicate statistically significant results.

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Figure S3: mean abundances of ground nesting (GN) in relation to wild boar foraging and shrub density. IkF(-) areas moderately foraged by wild boars; IkF(+) areas strongly foraged by wild boars. Black filled circles indicate areas with high density of shrubs; Open grey squares indicates areas with light shrub density. Lines indicate standard error of mean. Results are highly significant ($p < 10^{-6}$); see Table 2 and statistical analysis.

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602 was not statistically significant. Shrub cover was positively correlated to ground nesting birds
 603 abundances. The negative effect of lkf on birds abundances was marginally greater in S- than in S+
 604 areas, but this result was not statistically significant (Figure S3). Mean values of GN birds in
 605 relation to hypothesis decomposition are summarized in Table S2.

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609 **Table S2**
 610 **Mean values of GN birds in relation to hypothesis decomposition.**

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Abundances of GN birds and hypothesis decomposition

| Cell number | lkf | Tim | S | GN means | GN ± SE | GN -95% conf. Int. | GN +95% conf. Int. | N |
|-------------|-----|-------|-----|----------|---------|--------------------|--------------------|----|
| 1 | (-) | early | (+) | 5,4 | 0,4 | 4,6 | 6,3 | 7 |
| 2 | (-) | early | (-) | 3,5 | 0,5 | 2,4 | 4,6 | 4 |
| 3 | (-) | late | (+) | 6,7 | 0,6 | 5,4 | 7,9 | 3 |
| 4 | (-) | late | (-) | 5,1 | 0,3 | 4,4 | 5,8 | 11 |
| 5 | (+) | early | (+) | 2,8 | 0,5 | 1,7 | 3,8 | 4 |
| 6 | (+) | early | (-) | 1,2 | 0,3 | 0,5 | 1,8 | 11 |
| 7 | (+) | late | (+) | 4,5 | 0,4 | 3,6 | 5,4 | 6 |
| 8 | (+) | late | (-) | 3,0 | 0,4 | 2,1 | 3,9 | 6 |

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616 **Table S3**
 617 **Model selection for TN, GF and TF birds**

618 When two models had similar AIC and SS Residual scores, the most parsimonious model was
 619 chosen (see methods).

620

| Guild | selected variables | | | | | | | AIC | SS Residual |
|-----------------------------|--------------------|----------|----------|-----|----------|-----------|--------------|--------------|-------------|
| | lkf | Tim | S | Cut | Tim*S | lkf*Tim*S | | | |
| Tree Nesting (TN) | | | | | x | x | 220.9 | 225.5 | |
| | x | | | | | | 223.6 | 273.8 | |
| | | x | | | | | 222.9 | 269.3 | |
| | | | x | | | | 223.4 | 272.4 | |
| | | | | x | | | 223.7 | 274.4 | |
| Ground Foraging (GF) | | | | | x | x | 215.3 | 158.2 | |
| | | | | | x | | 215.8 | 171.6 | |
| | x | | | | | | 217.5 | 179.4 | |
| | | x | | | | | 218.7 | 184.9 | |
| | | | x | | | | 218.7 | 184.9 | |
| | | | | x | | | 218.7 | 185 | |
| Tree Foraging (TF) | x | x | x | | | | 169.5 | 77.9 | |
| | x | x | | | | | 169.2 | 82.7 | |
| | x | | | | | | 169.6 | 91 | |
| | | x | | | | | 173.5 | 98.6 | |
| | | | x | | | | 174.8 | 101.2 | |
| | | | | x | | | 175.3 | 102.4 | |

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625 We tested the effects of Ikf, Tim, S and Cut on the other guilds. Results of model selection are
626 summarized in Table S3. By contrast to ground nesting birds, abundances of birds belonging to tree
627 nesting (Figure 1b) or ground foraging (Figure 1c) guilds were not well explained by IkF or the
628 other studied variables; the observed changes were not statistically significant (Adjusted $R^2=0,05$;
629 $F=1,4$; $p=0,24$; Adjusted $R^2=0,01$; and $F=1,1$; $p=0,40$, respectively). The best model describing the
630 tree foraging birds abundances included Ikf and Tim as explanatory variables (Table S3), but the
631 observed changes were barely significant (Adjusted $R^2=0,12$; $F=2,0$; $p=0,07$). However, we noted
632 that tree foraging birds slightly tended to be more numerous in areas strongly foraged by wild boars,
633 a not significant result (Figure 1d).

634

635 We then tested the effects of Ikf, Tim, S and Cut on bird species diversity. The best model
636 describing species diversity included the variables Tim, IkF*S, Tim*S; Cut was rejected as an
637 explanatory variable (Table S4). The observed changes in species diversity were statistically
638 significant (Adjusted $R^2=0,33$; $F=4.6$; $p<10^{-3}$). We found that the main factor affecting bird species
639 diversity was the interaction of IkF*S (Figure 2, Table S5).

640

641 The present data indicate that, in accordance with H1, GN birds were less abundant in areas
642 strongly foraged by wild boars (Figure 1a). Thus, in accordance with H2, TN birds abundance was
643 not affected by wild boar foraging (Figure 1b). In addition, GF birds abundance was not correlated
644 to wild boar foraging (H3, Figure 1c).

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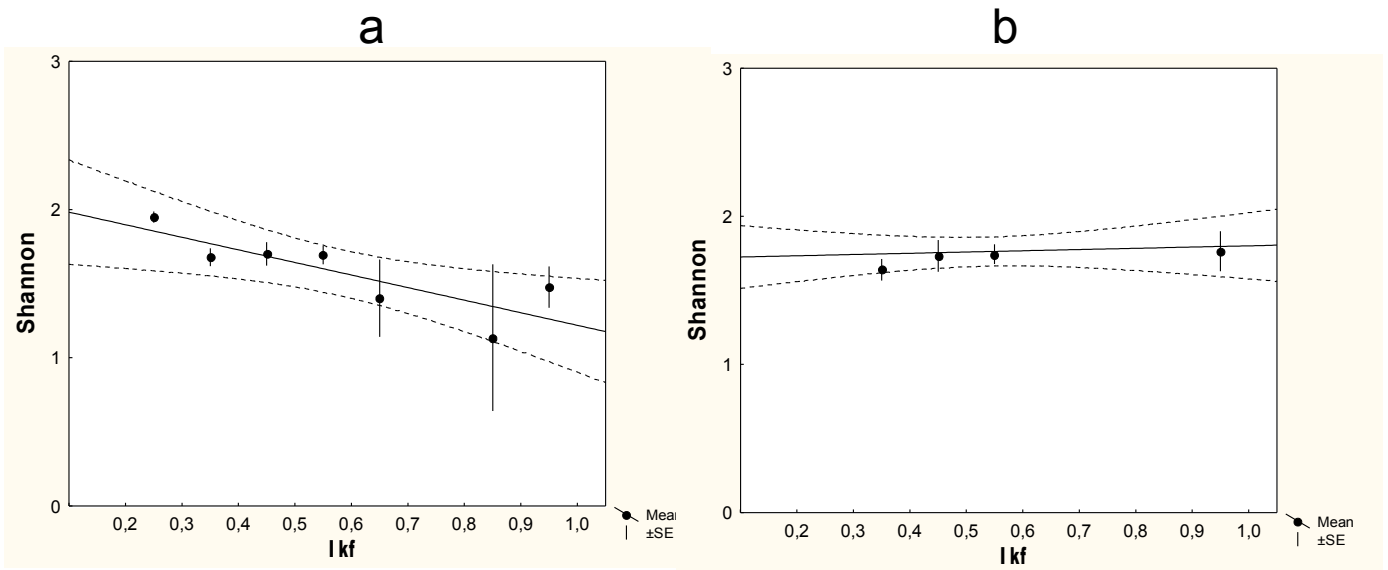


Figure 2: Variations of bird species diversity (Shannon) in relation to wild boar foraging (Ikf) and shrub density.

Predicted trends are presented as black lines with 95% confidence intervals (dashed lines). Bars indicate the standard error +mean. (a) bird species diversity in areas with low shrub cover. (b) bird species diversity in areas with dense shrub cover.

676

677 **Table S4**

678 **Model selection for species diversity**

679 When two models had similar AIC and SS Residual scores, the most parsimonious model was
 680 chosen (see methods).

681

682

| | lkf | Tim | selected variables | | lkf*S | Tim*S | AIC | SS Residual |
|--------------------------|-----|----------|--------------------|-----|----------|----------|-------------|-------------|
| | | | S | Cut | | | | |
| Species diversity | | x | | | x | x | 20,7 | 3,5 |
| | x | x | x | x | x | x | 21,7 | 3,5 |
| | | x | x | | | x | 25,1 | 4,2 |
| | | x | | | x | | 25,3 | 3,5 |
| | | | | | x | | 28,3 | 4,4 |
| | | x | | | | | 30,3 | 5 |
| | | | | | | x | 31,2 | 4,2 |
| | x | | | | | | 32,6 | 5,3 |
| | | | x | | | | 38,1 | 5,9 |
| | | | | x | | | 38,6 | 6 |

684

685 **Table S5**

686 **β coefficients and statistical parameters of the most parsimonious GLM model to explain**
 687 **species diversity using wild boar foraging traces (lkF), Timing (Tim) and shrub density (S) as**
 688 **predictors**

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Estimates (sigma-restricted parameterization)

| Effect | Level of effect | Column | Sp. div. param. | \pm SE | Sp. div. t | Sp. div. p | Sp. div. β | \pm SE |
|------------------|-----------------|----------|-----------------|------------|-----------------|-----------------|------------------|-------------|
| <i>Intercept</i> | | 1 | 1.7 | 0.0 | 38.49080 | 0.000000 | | |
| Tim | early | 2 | -0.1 | 0.0 | -1.92837 | 0.060277 | -0.24 | 0.13 |
| S | (+) | 3 | 0.1 | 0.0 | 1.27066 | 0.210526 | 0.16 | 0.12 |
| lkF | (-) | 4 | 0.1 | 0.0 | 1.33765 | 0.187885 | 0.17 | 0.13 |
| Tim*S | 1 | 5 | 0.1 | 0.0 | 1.91937 | 0.061437 | 0.24 | 0.13 |
| Tim*lkF | 1 | 6 | 0.0 | 0.0 | 0.63077 | 0.531456 | 0.08 | 0.13 |
| S*lkF | 1 | 7 | -0.1 | 0.0 | -2.34528 | 0.023587 | -0.30 | 0.13 |
| Tim*S*lkF | 1 | 8 | 0.0 | 0.0 | 0.51398 | 0.609837 | 0.06 | 0.12 |

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704 **4. Discussion and conclusions**

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706 **4.1. Wild boar foraging as an indicator of ecological change**

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708 Surprisingly, the use of foraging traces as an indicator of wild boar impact on ground nesting birds
709 has never been tested until now (Massei & Genov, 2004; Ballari & Barrios-Garcia, 2012). We found
710 that wild boar foraging traces provided a simple indicator of ground nesting birds abundance. Our
711 results evidenced the usefulness of this method since it was highly correlated with GN birds
712 abundances, but not with other avian guilds (tree nesting or ground foraging birds). The protocol
713 described in this study has numerous advantages. First, the measurements do not depend on
714 biological experts for birds species determination: in other words, anyone can perform an inventory
715 of wild boar foraging traces. Wild boar foraging traces are easily spotted, and the probability of
716 detection is likely constant among different observers using a standardized protocol. Second, wild
717 boar foraging traces are rapidly measured; this may prove useful for monitoring programs designed
718 to track annual changes of wild boar environmental impact on ground ecosystems at large spatial
719 scales. Third, measuring wild boar foraging traces is less climate dependent than measuring bird
720 abundances. Finally, the monitoring of wild boar impact on the avifauna do not requires the
721 previous knowledge of wild boar population size.

722

723 Understanding and measuring the environmental impact of wild ungulates in forest ecosystems
724 remains a challenge for researchers and wildlife managers. In practice, for management to succeed,
725 clear goals are required and the success of their application must be evaluated (Morellet *et al.*,
726 2007). The goal will often be a compromise such as maximising the revenue from hunting while
727 minimising damage to forestry/biodiversity (Williams *et al.*, 2002; Nichols & Williams, 2006).
728 Wildlife population trends are often assessed by species counts; surveillance monitoring frequently
729 requires several years to amass enough data to provide strong evidence of a decline in state before
730 action is taken. Species sampling methods remain labor intensive and time-consuming, and may be
731 subject to many biases (Kéry & Schmidt, 2008; Renwick *et al.*, 2012). As a result, there is an
732 increasing demand from policy makers to improve management by incorporating relevant indicators
733 that are simple, reliable and rapidly assessed. By recording a set of IECs, the aim is to achieve
734 management objectives, rather than exert a routine surveillance monitoring (Cederlund *et al.*, 1998;
735 Dale & Beyeler, 2001; Morellet *et al.*, 2007). To manage wild ungulates, it is thus suggested to track
736 over time the variations of at least three categories of indicators of ecological changes (Morellet *et al.*,
737 2007): one describing animal performance, another describing herbivore impact on
738 environment, and the last describing relative animal abundance.

739

740 As pointed by Dale and Beyeler (2001), the focus of ecological management programs becomes
741 narrow by selecting only one or few indicators, and may lead to oversimplified understanding of the
742 ecological interactions involved; the management of large ungulates is greatly improved by the
743 recording of a set of indicators (Morellet *et al.*, 2007). To avoid poorly informed management
744 decisions, indicators should take into account multiple levels in the ecological network in order to
745 effectively monitor the multiple levels of complexity of ecological systems (Dale & Beyeler, 2001;
746 Niemeijer & de Groot, 2008). Thus, a key challenge is to find a mix of measures which give easily
747 identified signals, can be used to track the ecological conditions at reasonable cost, and cover the
748 spectrum of ecological variation. The monitoring of our indicator of wild boar foraging responds to
749 these criteria; and thus managers dispose rapidly and easily of a diagnostic of wild boar potential
750 environmental impact on avifauna. However, if management interventions are required (*i.e.*, if the
751 manager estimates that wild boar impact on ground nesting birds may conflict with conservation
752 interests), the accuracy of management will be greatly improved by studies carried out to estimate
753 wild boar population size and trends. As such tools of wild boar abundance do already exist, based
754 on fecal drops (Acevedo *et al.*, 2007), one goal of this study (to complete the set of indicators for
755 wild boar management) was fulfilled. However, to be fully acknowledged as a new IEC, our

756 indicator should be ideally tested over a long period of time, in reference localities where true wild
757 boar population size is known (Dale & Beyeler, 2001; Morellet *et al.*, 2007). With a set of IECs, one
758 can envisage an adaptive management program (as defined by Nichols & Williams, 2006) of wild
759 boar game harvest compatible with bird conservation. The final step is then to set new hunting
760 quotas with the aim of provoking an adjustment in the population-habitat system in order to
761 approach the predefined goals. At least during the first years of monitoring, this is equivalent to a
762 trial-and-error process or adaptive management (Nichols & Williams, 2006; Morellet *et al.*, 2007).

763

764 **4.2. Effects of wild boar foraging on birds abundance**

765

766 To our knowledge, our study is the first to correlate bird abundances to an index based on the
767 frequency of wild boar foraging traces. Our results showed a significant decrease of 44,7% in
768 ground nesting birds abundances in areas strongly foraged by wild boars; in contrast, tree nesting
769 birds (out of reach of large ungulates) were not affected by wild boar gleaning. This result was
770 expected since ground nesting birds are highly impacted by other over-abundant ungulates
771 (Allombert *et al.*, 2005b; Holt *et al.*, 2011), and wild boars are known to be highly opportunistic
772 mammals that prey on anything near the ground: depredations on birds and nests (Saniga, 2002;
773 Saniga, 2003; Schaefer, 2004; Giménez-Anaya *et al.*, 2008; Carpio *et al.*, 2014b), reptiles and
774 amphibians (Jolley *et al.*, 2010), small mammals (Lozano *et al.*, 2007; Wilcox & van Vuren, 2009),
775 and other fauna and flora communities are substantial (see for review Barrios-Garcia & Ballari,
776 2012). The general assumption that all clutches of birds that nest on the ground represented
777 potential prey for wild boars is thus probably true. This predation effect may explain the observed
778 changes in bird distribution, as predation risk is known to influence prey-birds use of space
779 (Thomson *et al.*, 2006; Cresswell, 2008). We noted a cross-effect of wild boar foraging and habitat
780 on birds, *i.e.* the effects of wild boar foraging on species diversity were more important in areas
781 with low shrub cover. Other studies pointed that the density of potential nests and predation rates is
782 dependent on vegetation structure complexity and that foliage density near the nest may reduce
783 predation impact (Martin *et al.*, 1988; Chalfoun & Martin, 2009). In addition to resource
784 competition, predation risk is an important factor determining spatial distribution of birds (Suhonen,
785 1993) and offspring success (Martin, 1996; Thomson *et al.*, 2006; Chalfoun & Martin, 2007).
786 Interestingly, GN birds abundances were influenced by the timing of spot-count. As two of three
787 species composing the GN birds guild are long-migrant species, this result is not surprising. Further
788 research is needed to investigate the potential interactions between migration strategy and wild boar
789 impact, *i.e.*, if long-migrant birds are more impacted than partial-migrants.

790

791 We then explored the hypothesis that wild boar may compete with ground foraging birds for food
792 resource. Previous studies showed that wild boars negatively impacted rodents populations as a
793 result of direct competition for seeds (Focardi *et al.*, 2000) or legumes (Carpio *et al.*, 2014a).
794 Although wild boars are omnivorous and prey invertebrates, diet consist mainly of plant matter
795 including above-ground green material (Fournier-Chambrillon *et al.*, 1995; Baubet *et al.*, 2004).
796 The most obvious direct effect of rooting by wild boars is the reduction in plant cover (Singer *et al.*,
797 1984; Massei & Genov, 2004). Surprisingly, ground gleaning birds in our study were not
798 statistically affected by wild boar foraging. This strongly support the idea that wild boar mediated
799 effects observed in this study resulted mainly from direct predation on birds and nests or increased
800 predation risk, rather than food competition. In a remarkable long-term study, Wesolowski *et al.*
801 (2009) showed that wood warblers abundance in Bialowieza forest was inversely correlated with
802 small mammal predator densities and were only weakly correlated with caterpillar abundance; the
803 nomadic behavior of wood warblers appears to be a consequence of their attempts to breed in safe
804 places. Although most studies focused on the direct negative effects of wild ungulates on habitat
805 alterations and invertebrate availability, large herbivores may affect many species of plants and
806 animals through indirect effects (Suominen & Danell, 2006; Barrios-Garcia & Ballari, 2012). Large
807 ungulates concentrations may have both positive and negative effects on insectivorous birds;

808 positive effect being probably mediated by nutrient input through the ecosystem (Mathisen &
809 Skarpe, 2011; Mathisen *et al.*, 2012). This type of complex interaction may explain the observation
810 in our study, that tree foraging birds showed light increases in areas strongly foraged by wild boars.
811 However, one should note that ground nesting birds (Bonelli's warbler, that are also tree foraging
812 birds) were subtracted from this guild. As a result, this guild of “tree foraging birds” consisted of
813 only two species, which require a careful interpretation. Future research is thus needed to better
814 understand these mechanisms or confirm this result.

815

816 As final words, none of the birds species observed in this study are listed as threatened species;
817 however wild birds trends are commonly used as indicators of environmental health and
818 biodiversity (Gregory & van Strien, 2010). As common bird species shape the ecosystems, even
819 relatively small declines in their ranks can disrupt ecosystem structure (Gaston & Fuller, 2008); as a
820 consequence, monitoring bird indicators may give understanding on other taxas. Numerous studies
821 underlined the decline of common palearctic birds in Western Europe (Birdlife international, 2004;
822 Gregory & van Strien, 2010; EBCC, 2015). Recently, Gregory *et al.* (2007) showed that birds
823 nesting on the ground or in low vegetation declined more than those nesting higher up in shrubs and
824 trees. Species trends at the European scale are thus more significantly correlated to nest-type
825 (tree/shrub vs. ground/low vegetation) than to other ecological factors (migration, diet, region), a
826 report that is true for both farmlands and forest birds. It is also suggested that predation may be an
827 important factor influencing bird species trends in Europe (Fuller *et al.*, 2007b; Gregory *et al.*,
828 2007). Despite these reports, there are actually few managing programs that monitor the potential
829 impacts of wild boar at large scales. As wild boar populations will continue to grow in Europe
830 (Massei *et al.*, 2015), the development and use of new impact indicators may be critical for future
831 bird and environmental management.

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