1 **Landscape features associated to wind farms increase mammalian predator**

2 **abundance and ground-nest predation**

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12 **Abstract**

13 Wind farm implementation is a rapidly growing source of landscape transformation that may

14 alter ecological processes such as predator-prey interactions. We tested the hypothesis that wind

15 farms increase the activity of nest predators and, ultimately, increment ground-nest predation

16 rates. We placed 18 plots in Iberian shrub-steppes (11 at control and seven at wind farm sites),

17 each one comprised nine artificial ground-nests (three quail eggs/nest). Artificial nests were

18 placed during two events: at the beginning (April) and at the end (June) of the breeding season in

19 2016 (n = 324 artificial nests). We estimated the relative abundance of avian and large

20 mammalian predators in the surroundings of each plot and recorded nest fate after 12 days

21 exposure. We also measured variables at landscape and microhabitat scale that potentially affect

predator abundance and nest predation. Wind farm sites contained higher cover of gravel roads and more large mammalian predators. Moreover, the abundance of large mammalian predators increased with surrounding cover of both trees and gravel-roads. Avian predator abundance and nest predation rates did not differ between control and wind farm sites, though nest predation did increase with the surrounding cover of crops and gravel roads. Lastly, nest predation was higher at the end of the breeding season and decreased with moss and lichen cover. Our results support previous evidence on the increase of mammalian predator abundance as the surface area of gravel-roads increases, pointing towards a potential mechanism for wind farms leading to rise ground-nest predation. Future wind energy projects should minimize the development of gravel-roads for wind turbine access or maintenance.

Keywords: breeding success, gravel-roads, INLA-SPDE, mammalian predators, turbine access roads, wind energy

1. Introduction

The effects of wind farms on birds have been well documented (see e.g., Atienza et al. 2011; Erickson et al. 2014), with negative consequences on the abundance (De Lucas et al. 2004), occurrence (Pearce-Higgins et al. 2009; Stevens et al. 2013), space use (Farfán et al. 2009; Masden et al. 2009; Gómez-Catasús et al. 2019), and population trends (Campedelli et al. 2013; Gómez-Catasús et al. 2018a) of birds located in the vicinity of wind infrastructures. The direct effect of mortality through the collision with wind turbines and associated power lines has been widely recognized (Atienza et al. 2011; De Lucas et al. 2012; Erickson et al. 2014). In addition, disturbance due to visual, noise, or vibration impacts and barrier effects to movements have been described as potential sources of spatial displacement (Cabrera-Cruz and Villegas-Patraca 2016;

44   Zwart et al. 2016) with important consequences on individuals (e.g., through decrease of

45   survival, fecundity, or body condition; Martínez-Abraín et al. 2012; Winder et al. 2015) and

46   ultimately, on populations (Dahl et al., 2012). However, other indirect impacts such as the

47   potential influence of habitat alteration associated to wind farm implementation on ecological

48   processes (e.g., predator-prey interactions; Tylianakis et al. 2008) has been scarcely addressed

49   (Thaker et al. 2018), and the need of further research on this topic has been highlighted

50   (Rubenstahl et al. 2012; Hatchett et al. 2013; Gillespie and Dinsmore 2014; Smith and Dwyer

51   2016). Wind farm implementation could lead to changes in predator abundance, which in turn

52   could explain some indirect impacts of wind farms on avian survival and breeding success

53   (Smith and Dwyer 2016). Indeed, habitat changes associated with other human-made structures,

54   such as natural gas infrastructures, have been previously discussed as potential drivers of bird

55   population declines through the increase in nest predation rates (Hethcoat and Chalfoun 2015a),

56   among other mechanisms.

57   Landscape transformation associated with wind farm development may alter predation-prey

58   interactions by increasing the abundance of potential predators, or favouring habitat utilization

59   by predators. The installation of linear corridors, such as access gravel-roads to wind turbines,

60   could act as movement pathways for potential mammalian predators (Trombulak and Frissell

61   2000; Frey and Conover 2006; Van Der Ree et al. 2015), generating shifts in the structure and

62   composition of communities (e.g., augmentation of generalist predators; Degregorio et al. 2014;

63   Hethcoat and Chalfoun 2015a). Additionally, collision events with wind turbines might attract

64   avian and mammalian predators (Rogers et al. 2014) because of the higher availability of food

65   resources in their vicinity, as has been described from roadkill carcasses (Van Der Ree et al.

66   2015). Otherwise, a myriad of factors at landscape and microhabitat scale not related to wind

farm implementation may also affect habitat utilization by predators and nest predation rates. Landscape features, such as crops and forested areas, offer food and shelter for both mammalian and avian predators (Rodríguez-Pastor et al. 2016; Krüger et al. 2018), leading to higher predation rates in nests located closer to forest- or field-edges (Krüger et al. 2018; Ponce et al. 2018). Moreover, elevated structures such as trees are often used by avian predators as lookouts in search of prey (Widen 1994) and thus, nest survival may increase with distance from these structures (Ottvall et al. 2005). At the microhabitat scale, nest concealment increases with ground vegetation cover or height, negatively affecting nest predation rates (Whittingham and Evans 2004; Buehler et al. 2017; Ponce et al. 2018). Lastly, nest predation may vary among seasons (Evans et al., 2005; but see Calero-Riestra et al., 2013; Ponce et al., 2018), and these seasonal fluctuations might be associated with changes in predator activity or predator community (Hatchett et al. 2013).

The shrub-steppe is a crucial habitat for protecting avian biodiversity, as many shrub-steppe bird species have suffered long-term population declines across their range (Burfield 2005; BirdLife International 2015; Escandell 2017). Avian species linked to open environments, such as shrub-steppes, often nest on the ground as expected by the evolution of their nesting strategies and the structural simplicity of the landscapes they inhabit, making them prone to suffering high nest predation rates (Yanes and Suarez 1995). Indeed, predation has been described as the main cause of nest failure in open-land ground-nesting passerines (Calero-Riestra et al., 2013; Pérez-Granados et al., 2017; Ricklefs, 1969; Wright et al., 2009), and it is considered as a key factor limiting reproductive success and population viability (Martin 1993), especially in small populations (Vögeli et al. 2011). Therefore, open-land ground-nesting species maintain a fragile balance between reproductive success and predation risk, and increases in nest predation rates

caused by human-induced changes and local habitat disturbances (Suárez et al. 1993) may compromise the viability of open-land bird populations (Vögeli et al. 2011). Landscape transformations from wind farm development are growing sources of habitat degradation in shrub-steppes (Laiolo and Tella 2006), and slight increases in local predator abundance and nest predation rates in the vicinity of wind infrastructures might be a mechanism underlying the detected impacts on upland bird populations (Gómez-Catasús et al. 2018a; Fernández-Bellon et al. 2019).

In this study, we address the effect of wind farms on nest predator abundance and ground-nest predation rates in shrub-steppes. The specific objectives of our study were to: 1) assess the effect of wind farm occurrence on potential mammalian and avian nest-predator abundance, taking into account other landscape features that may influence habitat utilization by predators; and 2) evaluate the effect of wind farms on ground-nest predation through an artificial-nest predation experiment, controlling other factors measured at landscape (i.e., indices of predator abundance, landscape features, and proximity to sources of potential predators) and at microhabitat scale (i.e., plant structure) potentially affecting nest predation rates. We hypothesized that the development of turbine access gravel-roads associated with the implementation of wind farms would favour habitat utilization by predators (Van Der Ree et al. 2015). According to this hypothesis and the expected increase in food resources in the vicinity of wind turbines from collision fatalities, we predicted that the increase in nest-predator abundance would also lead to increased ground-nest predation rates. This work will allow to deepen our knowledge on indirect impacts of wind farms on wildlife, as well as disentangle the mechanisms explaining the effect of these infrastructures on the abundance, occurrence and trends of bird populations (De Lucas et al. 2004; Stevens et al. 2013; Gómez-Catasús et al. 2018a).

2. Methods and Materials

2.1 Study area

The study area is the "Tierra de Medinaceli" region located in the south of Soria province (central Spain; 2°26′35.1″W, 41°11′28.9″N; *c.* 1,200 m a.s.l.; Fig. 1) that covers around 200 km^2. The climate is Continental Mediterranean, with a mean temperature of 10.6°C and a mean annual rainfall of 500 mm. The landscape is a flat, short shrub steppe dominated by *Genista pumila*, *G. scorpius*, *Thymus* spp*.,* and *Linum spp.* (Garza et al. 2005). Cereal fields, ploughings, and conifer afforestations, subsidized by the Common Agricultural Policy (CAP) of the European Union, are interspersed. Bird community is composed by shrub-steppe birds nesting on the ground such as black-eared wheatear (*Oenanthe hispanica*), Dupont's lark (*Chersophilus duponti*), greater short-toed lark (*Calandrella brachydactyla*), red-legged partridge (*Alectoris rufa*)¸ thekla's lark (*Galerida theklae*), Eurasian skylark (*Alauda arvensis*), and tawny pipit (*Anthus campestris*), among others.

The Medinaceli Wind Resource Area is located in this region (Fig. 1) and it is composed of nine wind farms, each consisting of 10–32 turbines of 2000–2300 kW each. The landscape is fragmented as a result of natural (geological) processes and human activities, leading to shrub-steppes patches (see detailed information about study area in Gómez-Catasús et al. 2019). A total of 18 plots (ranging between one and two per shrub-steppe patch) were placed, 11 in the absence (hereafter control sites) and seven in the presence of wind farms (hereafter wind farm sites; Fig. 1). All plots were located 50 m from the nearest gravel-road (i.e., unpaved road surfaced with gravel) or path (i.e., track made by the transit of vehicles) to facilitate access, and were separated by a minimum distance of 1 km. Apart from the presence or absence of wind turbines, patches

135   with and without wind farms were of similar size (Mean ± SD; 135.06 ± 160.92 ha vs. 85.52 ±

136   59.52 ha in patches with and without wind farms, respectively; One-way ANOVA, F-value=

137   0.70, p = 0.42) and elevation (1164 ± 86 m vs. 1113 ± 58 m; One-way ANOVA, F-value = 1.75,

138   p = 0.21).

139   **2.2 Abundance of large mammalian and avian predators**

140   In May 2016 and along one 500-m transect per plot, we counted all scat belonging to large

141   mammals commonly described as nest predators in shrub-steppes (Pérez-Granados et al. 2017;

142   Suárez et al. 1993; Yanes and Suarez 1996). Number of scats was used as a proxy of large

143   mammalian predator abundance (Güthlin et al. 2014). Transects were conducted along the access

144   path or gravel-road to the plot, and they were centred in order to be representative of the whole

145   plot (Fig. 1). Transects were walked at constant speed and all scat found up to 1 m from both

146   sides of the observer were recorded and identified to species level (Bang et al. 2007). Scats that

147   could not be identified to species level were classified as *unknown*.

148   The abundance of avian predators was recorded from April to June 2016 using point-count

149   stations. Point-count stations were distributed over the study period, performing at least one

150   point-count station per month (April, May and June) in each plot. They were carried out by a

151   single observer, and under similar favourable weather conditions (i.e., absence of precipitation

152   and fog). Point-count stations (n = 188) were located both at the centre of each plot and

153   randomly spread throughout the study area, at distances from 68.3 up to 894.7 m to the centre of

154   the plot (mean ± SD = 475.2 ± 258.6 m). All potential avian nest predators (in accordance with

155   Suárez et al. 2009) sighted in a 500 m buffer around the point-count station were recorded, but

156   only those located within a 1 km radius around each plot were considered in subsequent

157   analyses. In case of the slightest indication that two observations belonged to the same

158   individual, one of them was discarded. Number of point-count stations in each plot ranged

159   between 9 and 40 (10.44 ± 3.31) and they extended between 20 and 410 mins (71.58 ± 79.07

160   mins), from sunrise to 2 p.m. Total effort time per plot ranged between 3.75 and 34.70 hours

161   (12.48 ± 11.20 h). Due to its random distribution, number of point-count stations located in the

162   surroundings of each plot and their duration differed between plots; therefore, the sampling effort

163   (i.e. number and duration of point-count stations) assigned to each plot was incorporated as an

164   offset in posterior analyses (see *2.5 Statistical analysis*).

165   **2.3 Nest predation experiment**

166   We placed nine artificial nests at each plot. Artificial nests were placed in a cross-shape and at 50

167   metres interval (Fig. 1). Under this layout, plots at wind farm sites were comprised of artificial

168   nests located in a distance gradient to wind turbines (Fig. 1). Artificial nests were placed in the

169   same location in two sampling periods in 2016: at the beginning (April) and at the end (June) of

170   the breeding season. Artificial nests were monitored after 12 days, which corresponds to the

171   average incubation period of ground-nesting species inhabiting in the study area (e.g., 11 days

172   Eurasian skylark, 12 days Dupont's lark, 14 days tawny pipit; Calero-Riestra et al. 2013; Pérez-

173   Granados et al. 2017; Suárez et al. 2009, 2005). Artificial nests were considered as predated

174   when at least one egg was either absent or damaged. Artificial nests damaged by ants were

175   discarded (n = 3) because they had holes that might attract predators or invalidate the artificial

176   nest. In addition, a whole plot (i.e., nine artificial nests) was discarded during the second

177   sampling period due to the placement of beehives in the surroundings, which may have affected

178   predator's behaviour. Thus, a total of 161 (99 at control and 62 at wind farm sites) and 151 (89 at

179   control and 62 at wind farm sites) artificial ground-nests were monitored during the first and

180   second sampling period, respectively (total of 312 artificial nests).

181   Artificial nests were placed that resembled local natural nests. They were laid on the ground

182   oriented towards N-NE (Yanes et al. 1996; Pérez-Granados et al. 2017) and next to a small

183   shrub, covering *ca.* 50% of the surface area occupied by the artificial nest (see Online Appendix

184   A). We tried to avoid differences in microhabitat variables in order to minimize their effect on

185   predation rates, although they were incorporated in the analysis in order to control for their

186   potential effect (see *2.4 Explanatory variables*). Artificial nests were placed in a scrape dug into

187   the ground with no lining, minimizing handling to avoid human odour that could attract

188   predators. Artificial nests were baited with three quail (*Coturnix coturnix*) eggs. All eggs had the

189   same origin, and they were handled under the same conditions (i.e. water-washed hands and

190   rubber protective gloves). One of the three eggs was emptied, filled with plaster, and tied to a

191   shrub with monofilament to avoid removal by predators and in order to potentially identify

192   predator class (i.e., large/small mammalian or avian) using the cues left at the plaster. Highly

193   disturbed artificial nests (i.e., the plaster egg was torn and detached from the monofilament)

194   where the vegetation in the surroundings was matted down and/or the soil removed, were

195   considered indicative of large mammalian or mesocarnivore predators. Remains of eggshell at

196   the artificial nest and marks from pecks on the plaster egg were considered indicative of bird

197   predation. Similarly, eggshell remains and small marks on the plaster egg clearly resembling the

198   teeth of a small rodent were considered indicative of small mammal predation. In the absence of

199   clear cues, potential nest predator was classified as *unknown*. Similar criteria for distinguishing

200   these classes of nest predators have been employed before (Best 1978; Yanes and Suarez 1996).

201   **2.4 Explanatory variables**

202   We characterized artificial nest locations at two spatial scales: 1) landscape scale, referring to the

203   habitat surrounding the artificial nest (i.e., measured in a 500-m radius-buffer around the centre

204   of each plot, hereafter plot level), and 2) microhabitat scale, referring to the immediate

205   surrounding of the artificial nest (i.e., measured at/from the artificial nest, hereafter nest level).

206   At the plot level, we considered wind farm occurrence, landscape features, and two indices of

207   predator abundance (Table 1). At the nest level, we incorporated the sampling period, the

208   proximity to sources of potential predators, and plant structure (Table 1).

209   *Wind farm occurrence* – Presence/absence of wind turbines in the shrub-steppe patch where plots

210   were located.

211   *Sampling period* – In order to control for potential seasonal patterns on nest predation rates

212   (Evans et al. 2005), we recorded the period at which the artificial nests were located (i.e., at the

213   beginning or at the end of the breeding season).

214   *Landscape features* – We characterized the landscape in a 500-m buffer around each plot (Fig.

215   1). We employed the Land Cover and Use Information System of Spain (SIOSE 2011) to

216   measure the surface area of the three major land-use types present in the study area: gravel-roads,

217   crops, and trees. These variables were calculated with the software QGIS 2.14.0 (Quantum GIS

218   Development Team 2019).

219   *Proximity to sources of potential predators* – We measured the distance from each artificial nest

220   to landscape structures that may be source of potential predators: distance to the nearest 1) tree;

221   2) crop; and 3) path or gravel-road. The distance to the nearest tree was estimated in the field

222   using a laser rangefinder (Leica 1200RF), whereas all the remaining variables were calculated in

223   QGIS 2.14.0 (Quantum GIS Development Team 2019).

224   *Indices of predator abundance* – Relative abundance of large mammalian (scat counts) and avian

225   predators per plot (see *2.2. Abundance of mammalian and avian predators*).

226 *Plant structure* – We carried out one vegetation sampling at each artificial nest location to

227 control for the potential effect of microhabitat on the probability of nest predation. Vegetation

228 structure was measured in 1x1 m quadrats to record nest concealment, placing the artificial nest

229 in the centre of the quadrat. We measured 13 vegetation structure variables describing cover

230 (horizontal structure) and architecture (vertical structure) at each artificial nest (see Rotenberry

231 and Wiens 1980 or Morales et al. 2008 for a similar approach). The importance of sampling both

232 structures (horizontal and vertical) relies on the effect of cover and height on nest concealment

233 and ultimately, on nest predation (Barrientos et al. 2009; Ponce et al. 2018). In each quadrat we

234 measured the following variables related with vertical plant structure: 1) maximum height; and

235 number of contacts at 2) 0-5 cm; 3) 5-10 cm; 4) 10-30 cm; and 5) above 30 cm height.

236 Additionally, we measured several descriptors of horizontal structure: 1) total vegetation cover

237 (%); 2) bare ground cover; 3) rock cover; 4) shrub cover; 5) perennial and annual herbaceous

238 cover; 6) detritus cover; 7) lichen cover; and 8) moss cover.

239 **2.5 Statistical analysis**

240 Fixed covariates were z-standardized (i.e., mean 0 and standard deviation 1) and tested for

241 collinearity prior to data analysis, retaining those predictors with a generalized variance inflation

242 factor (GVIF) lower than 2 (Fox and Monette 1992).

243 2.5.1. *Plant structure*

244 Regarding the variables describing plant structure (Table 1), we carried out Principal Component

245 Analysis (PCA) to reduce the number of predictors and to obtain interpretable gradients from an

246 ecological point of view. Two different PCAs were carried out incorporating those variables

247 related with: 1) vertical plant structure; and 2) horizontal plant structure. We retained those

248   components with an eigenvalue greater than 1. The function *prcomp* of the package "stats" from

249   the R software (v.3.6.2; R Core Team, 2019) was used.

250   2.5.2. *Relationship between wind farm occurrence and gravel-roads*

251   In order to identify whether gravel-roads were related to wind farm occurrence, we fitted two

252   linear models (Gaussian error distribution) to determine the best specification of random effects:

253   1) a spatial model, with a spatial random effect to control for potential spatial dependency (more

254   details below); and 2) a non-spatial model, without random effects. In both models, we

255   incorporated the surface area of gravel-roads as response variable and the factor 'wind farm

256   occurrence' as predictor. Spatial and non-spatial models were compared using the Watanabe-

257   Akaike information criterion (wAIC; Watanabe 2010), in which lower values means better fit.

258   We only present the results for the best model (lowest wAIC value), and we selected the spatial

259   model in case of similar wAIC values (i.e., ΔwAIC $\leq$ 2) because it produces more conservative

260   results (Zuur et al. 2017).

261   The spatial model was fitted to account for potential non-independence of data collected from

262   nearby plots. This was carried out by incorporating a spatial random effect using Integrated

263   Nested Laplace Approximation with Stochastic Partial Differential Equations (INLA-SPDE;

264   Lindgren et al. 2011). This method is an efficient and accurate way to model ecological data

265   while accounting for spatial and temporal dependencies (e.g., Myer et al., 2017; Ward et al.,

266   2015). Briefly, spatial dependency is accounted for using a latent Gaussian Markovian Random

267   Field (GMRF) and the covariance matrix of the spatial residual term is approximated via SPDE

268   framework. For that, we built a two-dimensional irregular grid (i.e., mesh) using a non-convex

269   boundary for the coordinates of all plots, and with a buffer zone in order to avoid edge effects for

270   the bordering vertices (Zuur et al. 2017) (see Online Appendix B). We used Penalized

271 Complexity priors (PC; Simpson et al. 2017) for the SPDE parameters to penalize the complexity

272 of the GMRF and thus, control flexibility, reduce over-fitting, and improve predictive

273 performance (Simpson et al. 2017; Fuglstad et al. 2019; Gómez-Rubio 2020). In accordance with

274 the study spatial dimension and the data, we set an uninformative PC-prior for the range (i.e., the

275 distance at which the spatial dependency diminish, r) so that $P(r < 7) = 0.5$, since 50% of the

276 distances between plots were *ca.* 7 km, and a PC-prior for the standard deviation where $P(\sigma >$

277 $3) = 0.05$.

278 2.5.3. *Predator abundance*

279 Due to the positive relationship between wind farms occurrence and the surface area of gravel-

280 roads (see *3. Results*) we assessed the effect of both predictors on predator abundance

281 independently. Thus, we carried out two independent analyses for each response variable (large

282 mammalian and avian predator abundance): 1) addressing the effect of wind farm occurrence per

283 se (factor 'wind farms occurrence' (presence/absence) as predictor); and 2) assessing the effect

284 of landscape features that may affect habitat use by predators (landscape predictors measured at

285 the plot level: surface area of gravel-roads, trees, and crops). In each set of analyses, we fitted a

286 spatial and a non-spatial Poisson model (count data, log-link function). Both models were

287 compared using wAIC to determine the best specification of random effects, and the best model

288 was selected following the same procedure explained in *2.5.2. Relationship between wind farm*

289 *occurrence and gravel-roads*. In the case of avian predator abundance, we incorporated an offset

290 to account for differences on sampling effort: the logarithm of the total time invested per plot.

291 Overdispersion was assessed simulating 1,000 datasets from the model and calculating the

292 dispersion statistic (Pearson residuals) for each simulated dataset $D|model$. Then, we compared

293 the dispersion statistic of the observed data $D|data$ with the distribution of the dispersion

294   statistic for the simulated data $D|model$. When $P(D|data > D|model)$ was either ≥ 0.975

295   (overdispersion) or ≤ 0.025 (underdispersion), the model was rejected.

296   2.5.4. *Probability of nest predation*

297   We split the set of predictors potentially affecting nest predation rates into three groups in order

298   to avoid model overparameterization (Table 2). These groups were defined on the basis of two-

299   fold criteria: 1) absence of collinearity among predictors; and 2) whether predictors were

300   descriptors of the context or predator abundance, landscape, or microhabitat. For each set of

301   predictors, we fitted a spatial and a non-spatial Bernoulli model (1 predated, 0 non-predated; log-

302   link function), which were compared using wAIC values to select the best partial model (see

303   *2.5.2. Relationship between wind farm occurrence and gravel-roads* for further explanation).

304   Lastly, we fitted two Bernoulli models (spatial and non-spatial) incorporating only those

305   predictors that showed an effect (i.e., 95% BCI for a given regression parameter did not contain

306   zero) on the probability of nest predation in each partial model. Spatial and non-spatial Bernoulli

307   models were compared by wAIC to select the best final Bernoulli model.

308   We fitted spatial models to account for non-independence of data collected from nearby artificial

309   nests using the INLA-SPDE approach (same PC-priors for the SPDE parameters than in *2.5.2.*

310   *Relationship between wind farm occurrence and gravel-roads*). We built a two-dimensional

311   irregular grid (i.e., mesh) using a non-convex boundary for the geographic coordinates of the

312   artificial nests and with a buffer zone in order to avoid edge effects for the bordering vertices

313   (Zuur et al. 2017) (see Online Appendix B). For the non-spatial model, we incorporated the

314   artificial nest ID nested within plot as a random intercept to account for multiple observations

315   from each artificial nest across plots (i.e., repeated-measures design, two sampling periods within

14

316 the same year). All models described above were fitted using the R package "INLA" (Rue et al.

317 2009) in the free R software (v.3.6.2; R Core Team 2019). We used INLA default prior

318 distributions for the intercept $\alpha \sim N(0,0)$, and the regression coefficients $\beta \sim N(0, 1000)$

319 (Gómez-Rubio 2020). Parameter estimates were reported as the posterior mean (β), associated

320 standard deviation (SD) and the 95% Bayesian Credible Interval (95% BCI). We considered

321 predictors to have an effect on the response variable when the parameter's 95% BCI did not

322 overlap zero (Zuur et al. 2017).

3. Results

3.1. Plant structure

325 The PCA on horizontal vegetation structure variables yielded four components (76.31% of total

326 variance) explaining 28.41%, 18.94%, 15.90%, and 13.07% of variance, respectively (see Online

327 Appendix C). The first component *PC1Hor* represented a gradient on vegetation cover since it

328 was positively associated with total vegetation cover (correlation coefficient = 0.85) and

329 herbaceous cover (0.76), and negatively with bare ground cover (-0.75). The second component

330 *PC2Hor* was positively associated with shrub cover (0.89) and negatively with rock cover (-

331 0.58). The third component *PC3Hor* was negatively related with detritus (-0.68) and moss cover

332 (-0.69), and the fourth component *PC4Hor* defined a negative gradient of lichen cover (-0.72).

333 On the other hand, the PCA on the vertical vegetation structure variables yielded two

334 components (62.89% of total variance, see Online Appendix C) which explained 41.63% and

335 21.27% of variance, respectively. The first component *PC1Ver* defined a positive gradient on the

336 number of contacts below 30 cm: number of contacts at 0-5 cm (0.67), 5-10 cm (0.77), and 10-30

337   cm (0.83). Lastly, the second component *PC2Ver* defined a negative gradient on maximum

338   vegetation height (-0.78).

339   *3.2. Relationship between wind farm occurrence and gravel-roads*

340   The spatial model performed better (wAIC= -106.7) than the non-spatial model (wAIC= 30.6;

341   Online Appendix D). The surface area of gravel-roads was higher at wind farm sites ($\beta \pm$ SD =

342   1.74 $\pm$; 95% BCI = [1.19; 2.31]; Fig. 2).

343   *3.3. Predator abundance*

344   Relative abundance of large mammalian predators (i.e., scat counts) ranged between zero and six

345   per plot (mean $\pm$ SD = 1.22 $\pm$ 2.02). Scats belonged to red fox (*Vulpes vulpes*; 68.2%), domestic

346   or feral dog (*Canis lupus familiaris*; 22.7%), and wild boar (*Sus scrofa*; 9.1%). The number of

347   avian predators recorded per hour ranged between zero and 0.74 (mean $\pm$ SD = 0.18 $\pm$ 0.22).

348   Potential avian predators registered were: carrion crow (*Corvus corone*; 48.8%), common raven

349   (*Corvus corax*; 21.9%), Eurasian magpie (*Pica pica*; 17.1%), and Montagu's harrier (*Circus*

350   *pygargus*; 12.2%).

351   Comparison of wAIC values indicated that spatial Poisson models performed better or

352   similarlythan non-spatial models (delta-wAIC between 0.01 and 27.7; Online Appendix D),

353   except for the model addressing the effect of landscape features on the relative abundance of

354   large mammalian predators for which the non-spatial model was selected (non-spatial model

355   wAIC= 69.9 and spatial model wAIC= 6.1 x 10^{17}). However, parameter estimates were similar

356   across both spatial and non-spatial models in all cases (see Online Appendix D). Poisson models

357   did not show indications of overdispersion (Online Appendix E). The relative abundance of large

358   mammalian predators was higher at wind farm sites ($\beta \pm$ SD= 2.33 $\pm$ 1.34; 95% BCI = [0.17;

16

359 5.51]) compared with control sites. However, the abundance of avian predators did not differ

360 between control and wind farm sites ($\beta \pm SD = 0.09 \pm 0.50$, BCI = [-0.93; 1.07]). Regarding the

361 effect of landscape features on predator abundance, the surface area of gravel-roads and trees had

362 a positive effect on the relative abundance of large mammalian predators (Fig. 3; Table 3),

363 whereas landscape variables did not influence the relative abundance of avian predators (Table

364 3).

365 *3.4. Probability of nest predation*

366 During the first survey at the beginning of the breeding season, a total of 22 out of 161 artificial

367 nests (13.6%) were predated, whereas 34 out of 151 artificial nests (22.5%) were predated at the

368 end of the breeding season (i.e., second sampling period). Thus, a total of 56 artificial nests were

369 predated during this experiment, which results in an average predation rate of 17.9%. Twenty-

370 seven out of the 56 artificial nests predated during the whole study period (48.2%) were located

371 at wind farm sites and 29 predated artificial nests (51.8%) were located at control sites.

372 Moreover, 38 artificial nests (67.9%) showed signs of large mammalian predation, five artificial

373 nests (8.9%) of bird predation, and five artificial nests (8.9%) of small mammal predation. Eight

374 artificial nests (14.3%) did not show clear cues and we were unable to identify predator species.

375 Comparison of wAIC values indicated that spatial Bernoulli models performed better or similarly

376 than non-spatial models. Thus, we present results from spatial Bernoulli models although

377 parameter estimates were similar across both models (see Online Appendix D). The final spatial

378 Bernoulli model incorporating all important predictors from the partial models (Table 4) showed

379 that the probability of nest predation differed between sampling periods, being higher at the end

380 of the breeding season (Table 5). Additionally, the probability of nest predation increased when

381 the cover of moss and detritus was lower around the nest (i.e., positive effect of *PC3Hor*; Table

382 5), and when the surface area of gravel-roads and crops increased (Table 5; Fig. 4). Wind farm

383 occurrence and the relative abundance of large mammalian and avian predators did not affect

384 nest predation (Table 4).

4. Discussion

386 Our results suggest that wind farms might increase nest predation rates through the increase in

387 relative abundance of large mammalian predators. Although wind farms did not directly increase

388 nest predation rates, we did see higher relative abundance of large mammalian predators there.

389 We also found that gravel-roads, which are more abundant on wind farms, were associated with

390 increased relative abundance of large mammalian predators and increased nest predation rates.

391 This connection points to a potential mechanism underlying the previously detected impact of

392 wind farms on population trends of an endangered shrub-steppe species in the study area, the

393 Dupont's lark (Gómez-Catasús et al. 2018a), since displacement or changes in nest-site selection

394 seems unlikely given the overall declining population trends experienced by the species (Gómez-

395 Catasús et al. 2018b), including those populations in the absence of wind farms (Gómez-Catasús

396 et al. 2018a). In this previous study, we observed that the Dupont's lark experienced a 21%

397 average annual decline in the presence of wind farms, *ca.* four times higher than in those

398 populations in the absence of wind infrastructures (5.8% average annual decline; Gómez-Catasús

399 et al. 2018a). The last finding of our research highlights the negative effect of crops interspersed

400 in natural steppe landscapes on ground-nest predation rates. This result, together with the effect

401 of gravel-roads, suggests that transformations of the shrub-steppe landscape may jeopardize

402 populations of bird species already in a critical conservation status.

Wind farm implementation is a growing source of landscape transformation in shrub-steppes (Laiolo and Tella 2006; Garza and Traba 2016), which in addition to introducing high turbines with important effects on visibility and bird and bat mortality (Erickson et al. 2014), increases the surface area of access-roads, as demonstrated in this work (Fig. 2). The effect of human-made structures and associated habitat modifications on the predator community has been previously described (Hethcoat and Chalfoun 2015a; Sanders and Chalfoun 2019), but it has been scarcely addressed for wind infrastructures (Thaker et al. 2018). In this work, the relative abundance of large mammalian predators was higher at wind farm sites, while avian predator abundance did not differ between control and wind farm sites. Habitat alteration associated to wind farm implementation is one potential mechanism explaining this effect because the abundance of large mammalian predators was positively associated with the surface area of gravel-roads around plots. This result is consistent with a wider body of evidence highlighting the role of roads as movement pathways for mammalian predators (Trombulak and Frissell 2000; Frey and Conover 2006; Van Der Ree et al. 2015), increasing the abundance of generalist predators (Degregorio et al. 2014; Hethcoat and Chalfoun 2015a). Another potential mechanism could be the attraction of mammalian predators to the surroundings of wind turbines due to the higher availability of food resources in the form of collision fatalities, similar to the attraction behaviour that has been described in some species from roadkill carcasses (Van Der Ree et al. 2015). The surface area of trees was also positively related with the relative abundance of large mammalian predators. Krüger et al. (2018) already highlighted the concentration of mammalian predator activity in areas closer to the forest edge. Among the predators identified in this study, the red fox and the wild boar show high ecological plasticity using mosaic landscapes that alternate shrub, forests, and crops (López-Martín 2010; Fernández-Llario 2017). Thus, forested

426   areas may offer shelter and food for mammalian predators, potentially explaining the higher

427   relative abundance of these predators in the surroundings.

428   Wind farms may cause avoidance behaviour in upland bird species (Pearce-Higgins et al. 2009),

429   and has been discussed as a potential driver of lower avian predator abundance or richness in the

430   surroundings of these infrastructures (Garvin et al. 2011; Keehn and Feldman 2018; Thaker et al.

431   2018). In this study, the abundance of avian predators was not affected by either the presence of

432   wind farms or by the landscape features in the surroundings. One possible explanation is that

433   control and wind farm plots were located relatively close (mean ± SD = 3.28 ± 0.92 km)

434   compared to home-range sizes of the avian species recorded, and hence both plots with and

435   without windfarms may fall within the home range of a single individual. Moreover, the species

436   forming the predator assemblage (corvids and harriers) in our study area may show complex and

437   species-specific responses (Chalfoun et al. 2002; Tewksbury et al. 2006), hindering detection of

438   a common effect on the avian predator community as a whole. The probability of collision and

439   thus avoidance rates at wind turbines depends on myriad factors such as species-specific flight

440   behaviour and morphology (Barrios and Rodríguez 2004; De Lucas et al. 2008). Some raptors

441   and less manoeuvrable species have shown clear patterns of avoidance of wind turbines (De

442   Lucas et al. 2008), but there are equivocal evidences about the effect of wind farms on the

443   collision risk and avoidance behaviour of corvids and harriers (Devereux et al. 2008; Wilson et

444   al. 2017; Bose et al. 2018), potential avian predators that co-occurred in our study area.

445   Similarly, the absence of effect of landscape features could be because avian predators present in

446   our study area differ in their habitat requirements or because the 500-m buffer that we used to

447   characterize the landscape may not be big enough to capture the habitat used for avian predators.

448   Therefore, local abundance of avian predators could not be explained through landscape features

449   or habitat changes induced by the implementation of wind turbines in shrub-steppes, and further

450   research is needed to address predator-specific responses. In any case, birds have been described

451   as uncommon nest predators in shrub-steppes (Calero-Riestra et al. 2013; Pérez-Granados et al.

452   2017; Yanes and Suarez 1996) and our results suggest that their density is low in the study area

453   (mean $\pm$ SD = 0.18 $\pm$ 0.22 avian predators per hour).

454   We found that the probability of nest predation was higher at the end of the breeding season

455   (June) as compared to the beginning of the breeding season (April). Our results are in agreement

456   with previous studies that also reported an increase on the probability of nest predation later in

457   the breeding season (Evans et al. 2005; Grant et al. 2005; but see Calero-Riestra et al. 2013).

458   This pattern might be associated with an increase on predator activity over the season (Hatchett

459   et al. 2013), due to a higher abundance of post-breeding adults and dispersal juveniles at the end

460   of the breeding season. Indeed, potential trade-offs have been described in multi-brooded

461   species, and specifically in ground-breeders, associated with early nesting at the beginning of the

462   breeding season even though optimal conditions have not been reached at that moment (i.e.,

463   lower food availability), compared with later broods at the end of the breeding season when food

464   availability is higher but also the risk of predation (Evans et al. 2005).

465   On the other hand, the probability of nest predation was not affected either by the occurrence or

466   the proximity to wind turbines. Previous studies did not detect an effect of wind turbines on nest

467   survival (Hatchett et al. 2013; Gillespie and Dinsmore 2014), while others detected an increase

468   on nesting success closer to turbines probably due to a lower activity of raptors in the vicinity of

469   wind farms (Rubenstahl et al. 2012). However, most of these studies did not incorporate

470   measures of landscape changes associated to wind energy development, which may be better

471   indices for quantifying the effects of wind energy development on nest predation rates. For

472 instance, Hethcoat and Chalfoun (2015b) found that in natural gas development facilities, well

473 density resulted in overly conservative estimates of nest survival probability, whereas predictors

474 quantifying landscape transformation (i.e., sagebrush habitat loss) negatively influenced nest

475 survival rates. Similarly, Mahoney and Chalfoun (2016) reported an increase on nesting success

476 as turbine density decreased. In accordance with these results, we did not detect a direct effect of

477 wind turbine occurrence on the probability of ground-nest predation, but we did find evidence

478 for a positive effect of the amount of gravel-roads in the surroundings, a landscape

479 transformation associated with wind farm implementation (Fig. 2). Indeed, the probability of nest

480 predation drastically increased above 2 ha of gravel-roads (i.e., 2.5% of the surface area covered

481 by the 500 m buffer around each plot at which the landscape was characterized; Fig. 4) and a

482 25% predation probability was reached at 4 ha (i.e., 5% of the total surface area; Fig. 4),

483 suggesting that small proportions of this landscape feature can compromise the breeding success

484 of ground-nest breeders.

485 Consistent with our predictions, the relative abundance of large mammalian predators and the

486 probability of nest predation were higher in shrub-steppe patches with a higher surface area of

487 gravel-roads. However, no direct effect of predator abundance indices, either avian or large

488 mammalian, on the probability of nest predation was detected. One possible explanation is that

489 the samplings of the relative abundance of large mammalian (late May) and avian predators

490 (from April to June) are decoupled from each specific nest predation experiment (April and June,

491 respectively), hindering to detect an effect on the probability of nest predation. Moreover, nest

492 predation could depend on the activity of a particular predator species and not on the whole

493 predator assemblage (Benson et al. 2010; Hethcoat and Chalfoun 2015a). Lastly, one pitfall of

494 our study design is that abundance of other potential predators, also linked to roads, was not

495 properly surveyed. This is the case for small mammals or reptiles, which have been recorded as

496 potential predators of our artificial nests (8.9% of the artificial nests for small mammals) and

497 described as common nest predators in shrub-steppes (garden dormouse *Elyomis quercinus*,

498 ocellated lizard *Timon Lepidus*, and lataste viper *Vipera latastei*; Pérez-Granados et al. 2017;

499 Suárez et al. 1993; Yanes and Suarez 1996). Roadside verges have been described as refuges for

500 small mammals (Ruiz-Capillas et al. 2013) and are used by reptiles for thermoregulation (Ashley

501 and Robinson 1996). Indeed, other studies have reported cascade impacts of wind farms on the

502 density of ground-dwelling prey (e.g., lizards) as a consequence of the lower abundance and

503 activity of predatory birds (Thaker et al. 2018). Therefore, future studies should address the

504 effect of wind farms on this predator community and its role on ground-nest predation.

505 The expansion of crops as a result of agricultural intensification is one of the major sources of

506 habitat loss and fragmentation on shrub-steppe ecosystems (Laiolo and Tella 2006) that might

507 affect nest predation in several non-mutually ways (e.g., increasing predator abundance or nest

508 densities in suboptimal habitats; Evans, 2004). In this study, we found a positive association

509 between crop surface area and the probability of nest predation. The role of non-cropped habitats

510 (e.g. field margins) as undisturbed and stable refuges for mammals and other potential predators

511 is well-known (Benton et al. 2003; Rodríguez-Pastor et al. 2016) thus increasing predation risk

512 on nests located closer to agricultural edges (Ponce et al. 2018). Similarly, stone walls used to

513 delimit crop fields in our study area are commonly used by lacertids and other reptiles (Díaz et

514 al. 2006). Therefore, it is reasonable that the probability of nest predation increases with the

515 surface area of crops, which might be an important source of potential predators. In particular,

516 the probability of nest predation was null or low when the surface area of crops in the

517 surroundings was lower than 20 ha (i.e., *ca.* 25% of the total surface area characterised in the

518   surroundings within the 500 m buffer; Fig. 4), and it greatly increased when the surface area of

519   crops was above 20 ha, reaching *ca.* a 50% predation probability when the surface area of crops

520   was above 40 ha (i.e., *ca.* 50% of the total surface area; Fig. 4). Thus, the effect of crops on

521   predation rates might be specially worrying if crop amount is above 25% of the total surface area

522   in the surroundings. Lastly, microhabitat variables such as vegetation cover and height, also

523   influence nest predation rates (Barrientos et al. 2009; Praus and Weidinger 2015; Ponce et al.

524   2018). In this study, the probability of nest predation decreased with the cover of moss and

525   detritus, whereas the remaining microhabitat variables did not determine nest predation rates. We

526   attempted to minimize the differences on microhabitat variables when placing the artificial nests

527   (i.e., under a short shrub covering *ca.* 50% of the nest). Therefore, it is reasonable that most of

528   the microhabitat variables related with plant structure did not influence nest predation rates. One

529   possible explanation for the effect of moss and lichen cover on predation rates is that they may

530   offer better nest concealment and reduce nest conspicuousness, thus reducing nest predation

531   (Hatchwell et al. 1996).

532   The use of artificial nests to answer ecological questions is under debate (see review in Moore

533   and Robinson 2004). The degree to which results from artificial nests can be generalized to real

534   bird populations has been questioned because artificial nests might attract different predators

535   than real nests and hence, may be predated at different rates (Thompson and Burhans 2004). Nest

536   predation rates in our study (17.9%) were similar than predation rates previously described in our

537   study area from natural nests during the incubation period (18.9%, n = 27; Pérez-Granados et al.

538   2017). Moreover, the predation rate obtained after monitoring 27 nests of black-eared wheatear

539   (3), Dupont's lark (7), greater short-toed lark (6), Eurasian skylark (10) and tawny pipit (1)

540   during the incubation period in the study area (years 2017-2019) was 33.3% (authors'

541 unpublished data). In any case, neither breeding success nor overall predation rates were within

542 the aims of this study. A drawback of our study is that nest predators might not be reliably

543 assigned based on cues left at artificial nests (Williams and Wood 2002; Thompson and Burhans

544 2004) and thus, these results may be misinterpreting the importance of some predators (Benson

545 et al. 2010). Our goal was to identify main nest predators in order to contextualize factors at

546 microhabitat and landscape scales determining the probability of nest predation. We consider

547 that we succeeded in this purpose since mammals have previously been described as main

548 predators from natural nests in shrub-steppes, whereas reptiles, birds and small mammals were

549 considered as rare (Suárez et al. 1993). Therefore, while our results cannot be used to obtain

550 predation rates at natural nests, we consider that they provide important insights on spatial trends

551 in predation that may have implications for natural nests. Moreover, some authors admit that

552 they provide useful data with larger sample sizes as compared to those from natural nests due to

553 the time-consuming task of finding real nests, highlighting their utility for comparative purposes

554 (Major and Kendal 1996; Evans 2003) and as indices of relative predation pressure (Vögeli et al.

555 2011). Thus, main conclusions drawn from this study about the role of gravel-roads and crops as

556 sources of predators remain reliable.

557 **5. Conclusions and implications for conservation**

558 This paper delves into a potential mechanism explaining the negative effects of wind farms on

559 ground-nesting birds: the increase of large mammalian predator abundance through landscape

560 transformation that may lead to higher ground-nest predation rates. Specifically, this indirect

561 impact has been addressed in shrub-steppes, a key ecosystem for the preservation of avian

562 wildlife since shrub-steppe birds experience declining population trends (Burfield 2005;

563   Sanderson et al. 2005), which might be aggravated in the presence of wind infrastructures

564   (Gómez-Catasús et al. 2018a). Future research should address the effects of wind farms and

565   landscape features on avian breeding success monitoring real nests throughout all stages of the

566   nesting cycle, as well as studying patterns of nest-site selection or the effect of turbine noise on

567   predator-prey interactions.

568   Predator control has been widely proposed as an effective conservation measure of vulnerable

569   bird populations (Smith et al. 2010). However, a deeper understanding of the mechanisms that

570   relate wind farm occurrence, habitat, and landscape to breeding success and other predators not

571   specifically covered by this study (e.g., small mammals and reptiles) is needed to determine

572   whether these management practices could be effective in shrub-steppe ecosystems. Meanwhile,

573   managers should prioritize alternative solutions such as installing narrow and straight access-

574   roads or reusing already-existing roads, along with placing wind turbines outside the distribution

575   area of endangered bird species. These specific recommendations aim to minimize the

576   development of gravel-roads for access and maintenance of wind turbines during the design of

577   future wind energy projects, in order to avoid shifts on the predator community and the potential

578   increase on predation rates.

Acknowledgments

Electronic supplementary material

583 **Appendix A.** Artificial ground-nests
584 **Appendix B.** Meshes
585 **Appendix C.** Results of Principal Component Analyses
586 **Appendix D.** Parameters estimates and wAIC values for spatial and non-spatial models
587 **Appendix E.** Model validation – Overdispersion assessment

Declarations

594   **Conflicts of interest/Competing interests**

595   None

596   **Ethics approval** (include appropriate approvals or waivers)

597   Not applicable

598   **Consent to participate** (include appropriate statements)

599   Not applicable

600   **Consent for publication** (include appropriate statements)

601   Not applicable

602   **Availability of data and material** (data transparency)

603   Data will be submitted to an appropriate public data repository

604   **Code availability** (software application or custom code)

605   R-code will be submitted to an appropriate public repository

606   **Author contributions**

607   **Julia Gómez-Catasús**: Conceptualization, Methodology, Formal Analysis, Investigation and
608   Writing - Original Draft, Visualization. **Adrián Barrero**: Conceptualization, Methodology,
609   Investigation and Writing - Review & Editing. **Margarita Reverter**: Investigation and Writing -
610   Review & Editing. **Daniel Bustillo-de la Rosa**: Investigation and Writing - Review & Editing.
611   **Cristian Pérez-Granados**: Investigation and Writing - Review & Editing. **Juan Traba**: Writing
612   - Review & Editing, Supervision, Project administration and Funding acquisition.

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TABLES

898	**Table 1.** Explanatory variables incorporated in the analyses carried out in this study. The level (nest or

899	plot) at which the different variables were measured is indicated. In addition, the analysis in which the

900	explanatory variables were incorporated as predictors is stated. PA: Predator abundance. NP: Nest

901	predation

Environmental factor	Variables	Level	Analysis
Wind farm occurrence	Presence/Absence of wind turbines	Plot	PA, NP
Sampling period	Beginning or end of the breeding season	Nest	NP
Landscape features	Surface area of gravel-roads (m^2) Surface area of crops (m^2) Surface area of trees (m^2)	Plot - 500 m buffer	PA, NP
Proximity to sources of potential predators	Distance to the nearest path/gravel-road (m) Distance to the nearest crop (m) Distance to the nearest tree (m)	Nest	NP
Indices of predator abundance	Large mammalian predators Avian predators	Plot – 500 m transect (large mammalian predators) and 1 km buffer (avian predators)	NP
Plant structure	Vertical structure: Maximum height (cm) Number of contacts at 0-5cm, 5-10cm, 10-30cm and >30 cm Horizontal structure: Total vegetation cover (%) Bare ground cover (%) Rock cover (%) Shrub cover (%) Herbaceous cover (%) Detritus cover (%) Lichen cover (%) Moss cover (%)	Nest - 1x1 m quadrat	NP

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905 **Table 2.** Partial Bernoulli models fitted to address the factors explaining the probability of nest predation

906 at 312 artificial nests placed in two occasions (April and June) in 2016. Predictors incorporated in each

907 partial model are indicated.

Model	Predictors
Context and indices of predator abundance	Period (beginning or end of the breeding season) Wind farm occurrence (presence/absence) Indices of predator abundance (large mammalian and avian predators)
Landscape	Surface area of gravel-roads, trees, and crops (plot level) Distance to the nearest path/gravel-road, tree, and crop (nest level)
Plant structure	PC1Hor, PC2Hor, PC3Hor, PC4Hor PC1Ver, PC2Ver [a]

908 [a] PC1Hor-PC4Hor states for the Principal Components yielded by the PCA on the horizontal plant structure,

909 whereas PC1Ver-PC2Ver are the Principal Components yielded by the PCA on the vertical plant structure (see

910 *Results*)

911 **Table 3.** Results of the non-spatial Poisson model assessing the effect of landscape features on the

912 relative abundance of large mammalian predators and the spatial Poisson model assessing the effect of

913 landscape features on the relative abundance of avian predators (n= 18 plots in 2016). Posterior mean (β),

914 standard deviation (SD) and 95% Bayesian Credible intervals (95% BCI) are shown for the intercept and

915 each landscape variable. Predictors with an important effect on the relative abundance of large

916 mammalian predators are highlighted in bold.

	Mammalian predators			Avian predators		
	β	SD	95% BCI	β	SD	95% BCI
Intercept	-0.25	0.29	[-0.86; 0.27]	-1.88	1.15	[-3.41; -0.69]
Surface area of roads	**0.63**	**0.18**	**[0.28; 1.00]**	0.16	0.28	[-0.42; 0.73]
Surface area of trees	**0.52**	**0.23**	**[0.07; 0.96]**	-0.10	0.26	[-0.61; 0.44]
Surface area of crops	-0.40	0.35	[-1.14; 0.22]	0.10	0.29	[-0.49; 0.70]

917

918   **Table 4.** Results of the three partial spatial-Bernoulli models addressing the factors explaining the

919   probability of nest predation at 312 artificial nests placed in two occasions (April and June) in 2016. The

920   Posterior mean (β), standard deviation (SD), and 95% Bayesian Credible Intervals (95% BCI) are shown

921   for the intercept and each variable independently for each partial model. Predictors with an important

922   effect on the probability of nest predation are highlighted in bold.

| Model		β	SD	95% BCI
Context	Intercept	-3.43	1.01	[-5.66; -1.64]
	Wind farm occurrence (Presence)	1.41	1.39	[-1.19; 4.37]
	Period (End breeding season)	**0.73**	**0.34**	**[0.07; 1.41]**
	Abundance of avian predators	0.55	0.54	[-0.44; 1.71]
	Abundance of mammalian predators	-0.33	0.62	[-1.63; 0.87]
Landscape	Intercept	-2.58	0.76	[-4.30; -1.32]
	Surface area of gravel-roads	**0.99**	**0.46**	**[0.17; 2.01]**
	Surface area of crops	**1.33**	**0.48**	**[0.48; 2.37]**
	Surface area of trees	-0.85	0.50	[-1.97; 0.01]
	Distance to nearest path/gravel-road	0.12	0.23	[-0.33; 0.56]
	Distance to nearest tree	-0.16	0.24	[-0.65; 0.32]
	Distance to nearest crop	0.31	0.34	[-0.35; 0.98]
Microhabitat [a]	Intercept	-2.54	0.63	[-3.92; -1.43]
	PC1Hor	0.44	0.24	[-0.01; 0.92]
	PC2Hor	0.32	0.23	[-0.11; 0.78]
	PC3Hor	**0.73**	**0.27**	**[0.22; 1.30]**
	PC4Hor	0.31	0.20	[-0.07; 0.71]
	PC1Ver	0.89	0.19	[-0.29; 0.45]
	PC2Ver	0.16	0.20	[-0.22; 0.55]

923

924 **Table 5.** Results of the final spatial-Bernoulli model addressing the factors explaining the probability of

925 nest predation at 312 artificial nests placed in two occasions (April and June) in 2016. The Posterior mean

926 (β), standard deviation (SD) and 95% Bayesian Credible intervals (95% BCI) are shown for the intercept

927 and each variable. Predictors with an effect on the probability of nest predation are highlighted in bold.

	β	SD	95% BCI
Intercept	-2.74	0.62	[-4.09; -1.64]
Period (End breeding season)	**0.73**	**0.34**	**[0.07; 1.42]**
Surface area of gravel-roads	**0.76**	**0.39**	**[0.04; 1.60]**
Surface area of crops	**0.94**	**0.41**	**[0.20; 1.83]**
PC3hor[a]	**0.56**	**0.24**	**[0.09; 1.05]**

928
929 [a] Third Principal Component yielded by the PCA on horizontal vegetation structure variables. It was negatively related with detritus and moss cover.

930

931   **FIGURES**

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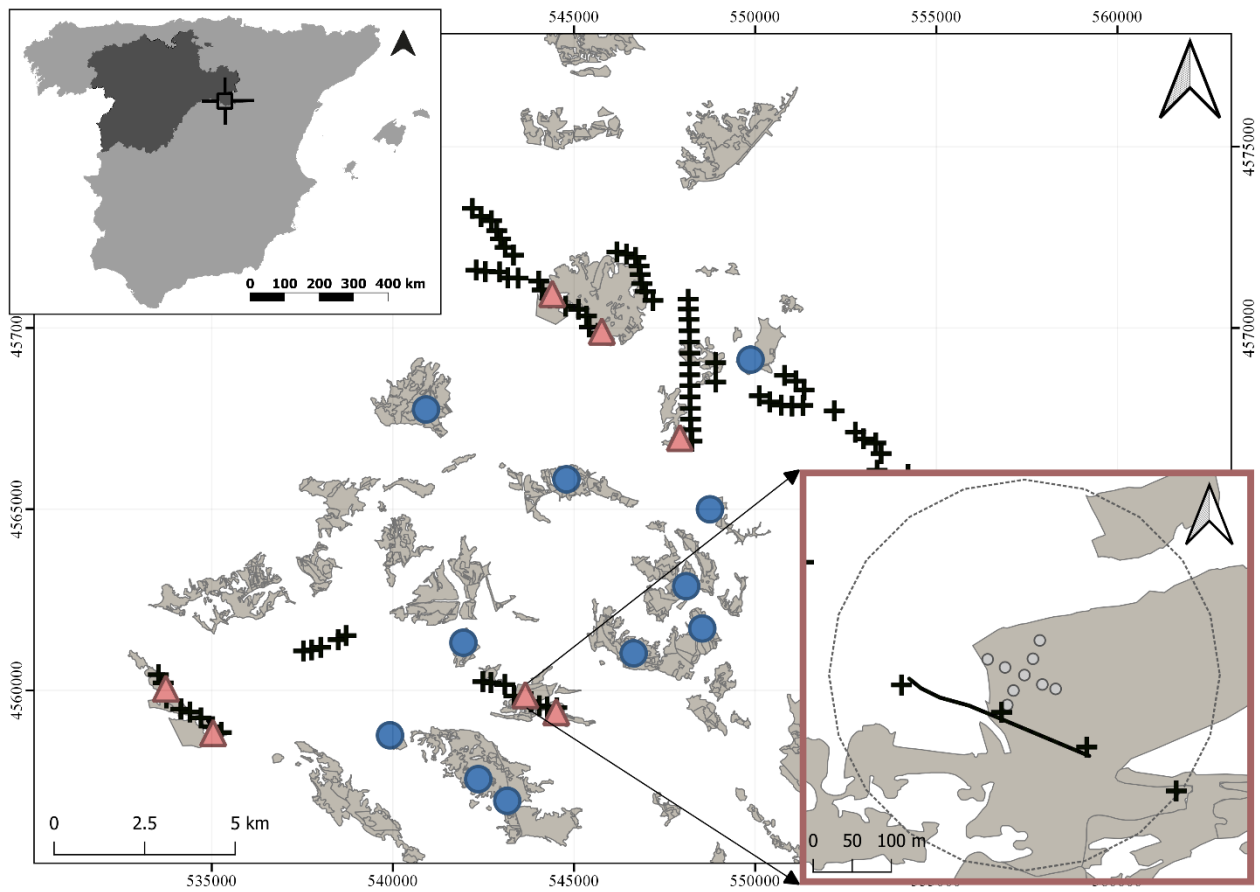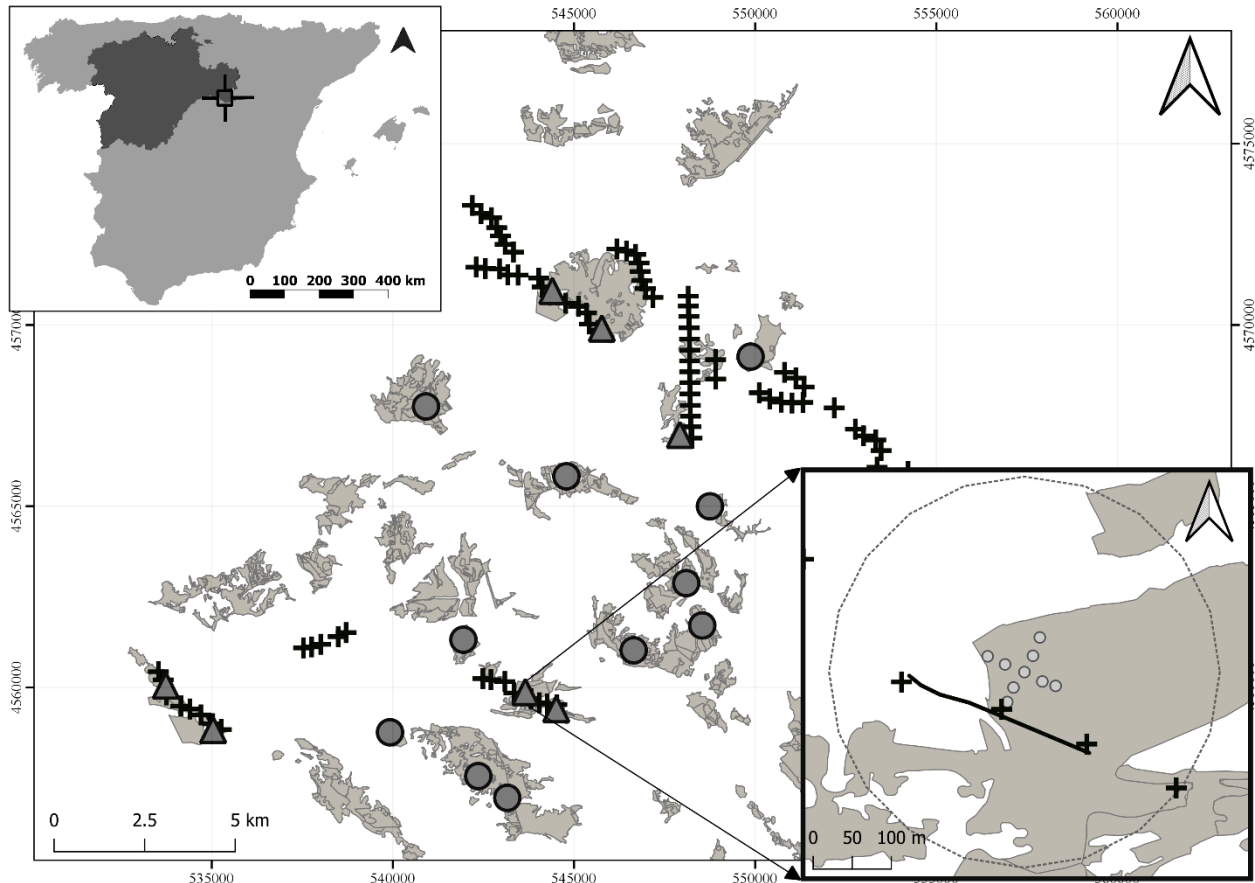
Fig. 1 Map of the study area. Plots at control (circles; n = 11) and wind farm sites (triangles; n = 7) are

depicted. Shrub-steppe patches are delimited and the location of wind turbines (black crosses) are

represented. At bottom right, one plot is expanded, depicting the layout of the nine artificial ground-nests

(small grey circles). In addition, the walked transects to estimate the relative abundance of large

mammalian predators (black line) and the 500 m buffer where landscape features were measured (grey

circle) are also depicted. The insect shows the location of the study area (black rectangle) in Spain.

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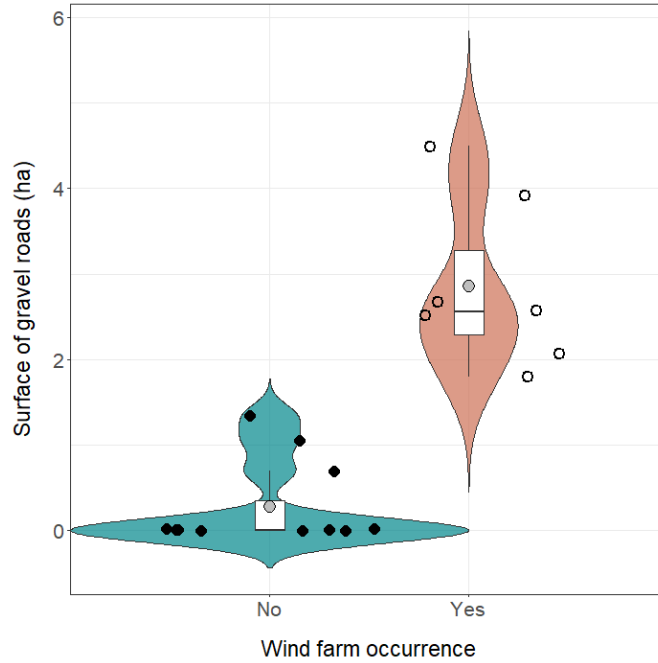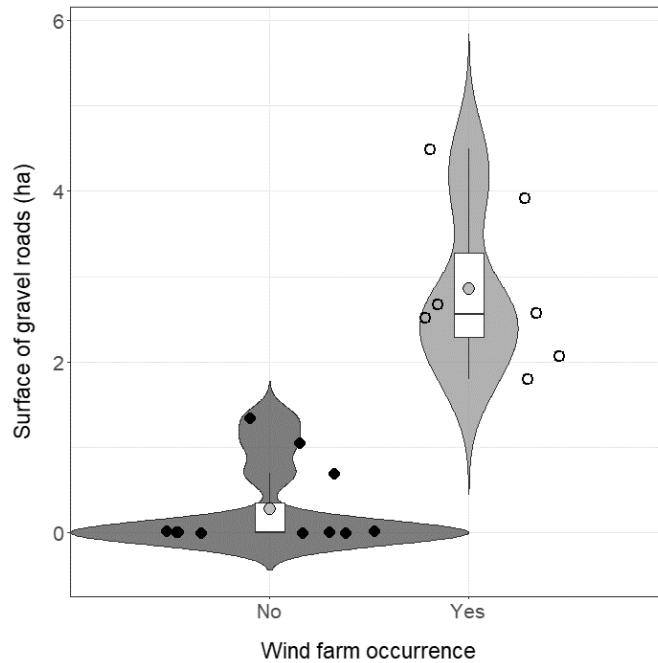
Fig. 2 Surface area of gravel-roads (ha) in plots at control (n = 11; filled dots) and wind farm sites (n = 7;

empty dots). Median (black line), mean (grey dot), and 25th and 75th percentiles (box) are depicted. The

violin plot (surfaces) represents the distribution (kernel probability density) of the surface of gravel roads

predicted by the model at control and wind farm sites.

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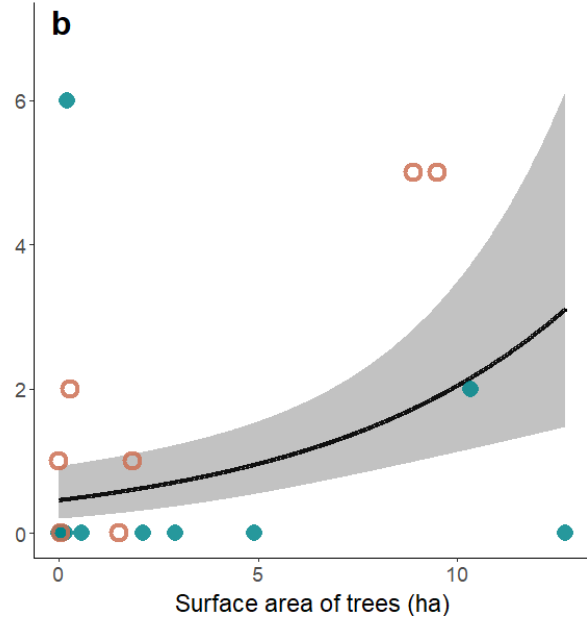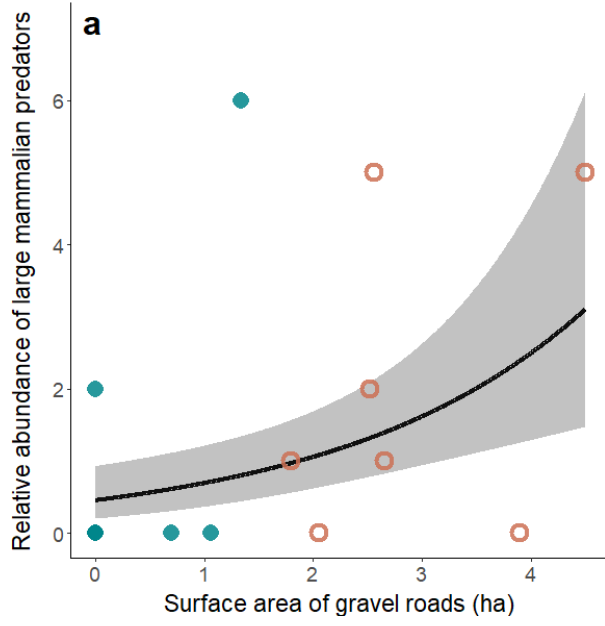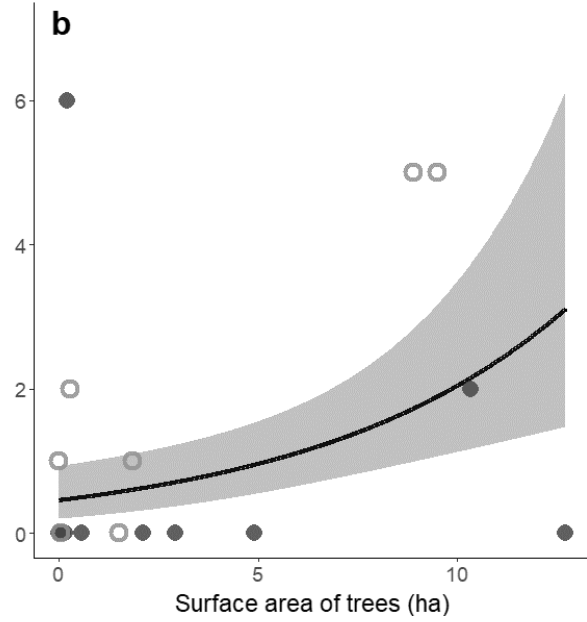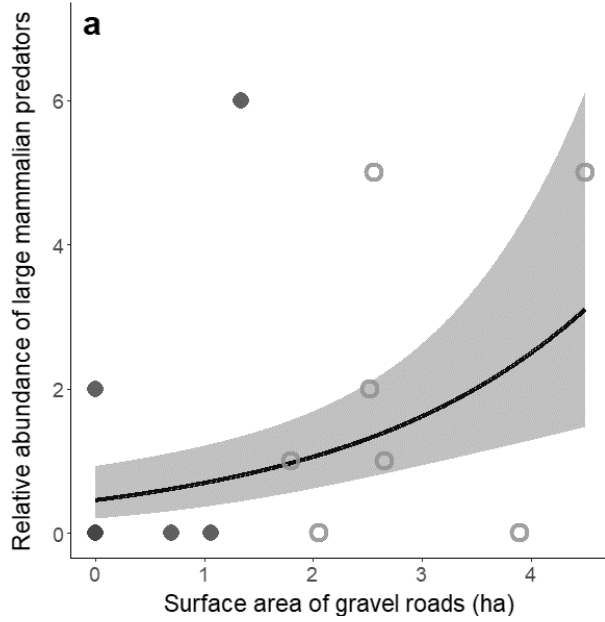
953

954 **Fig. 3** Relative abundance of large mammalian predators (i.e., scat counts) in relation to: (a) the surface

955 area of gravel-roads (ha); and (b) the surface area of trees (ha) in a 500 metres (radius) buffer around each

956 plot. Observed values for the plots located at control (n = 11; filled dots) and wind farm sites (n = 7;

957 empty dots) are depicted. Mean (black line) and 95% Bayesian Credible Interval (grey surface) are

958 represented.

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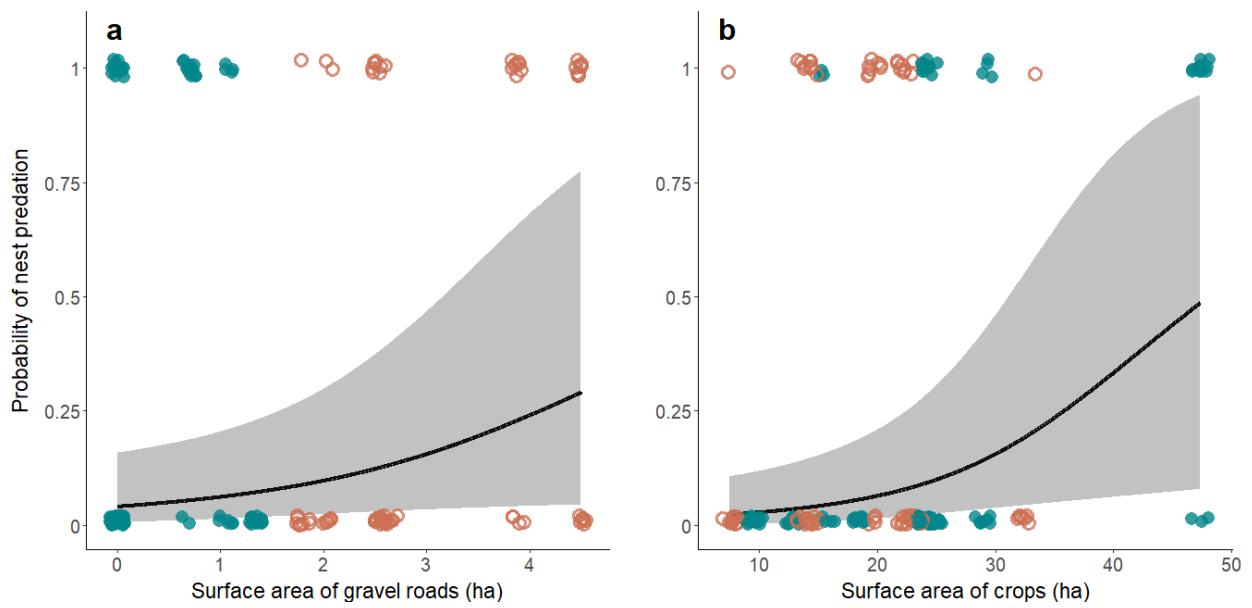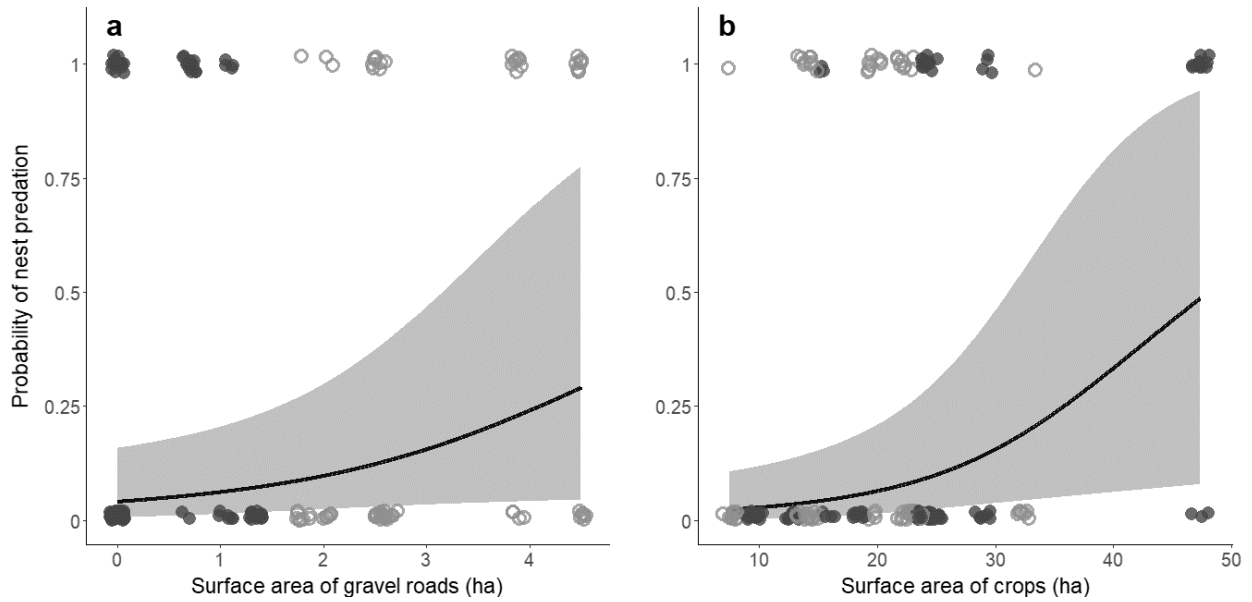
Fig. 4 Probability of nest predation in relation to: (a) the surface area of gravel-roads (ha); and (b) the

surface area of crops (ha) in a 500 metres (radius) buffer around each plot. Observed values for the

artificial nests located at control (n = 188; filled dots) and wind farm sites (n = 124; empty dots) are

depicted. Lastly, mean (black line) and 95% Bayesian Credible Interval (grey surface) are represented.

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