Landscape features associated to wind farms increase mammalian predator abundance and ground-nest predation

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Abstract

Wind farm implementation is a rapidly growing source of landscape transformation that may alter ecological processes such as predator-prey interactions. We tested the hypothesis that wind farms increase the activity of nest predators and, ultimately, increment ground-nest predation rates. We placed 18 plots in Iberian shrub-steppes (11 at control and seven at wind farm sites), each one comprised nine artificial ground-nests (three quail eggs/nest). Artificial nests were placed during two events: at the beginning (April) and at the end (June) of the breeding season in 2016 (n = 324 artificial nests). We estimated the relative abundance of avian and large mammalian predators in the surroundings of each plot and recorded nest fate after 12 days 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;
Zwart et al. 2016) with important consequences on individuals (e.g., through decrease of
survival, fecundity, or body condition; Martínez-Abraín et al. 2012; Winder et al. 2015) and
ultimately, on populations (Dahl et al., 2012). However, other indirect impacts such as the
potential influence of habitat alteration associated to wind farm implementation on ecological
processes (e.g., predator-prey interactions; Tylianakis et al. 2008) has been scarcely addressed
(Thaker et al. 2018), and the need of further research on this topic has been highlighted
(Rubenstahl et al. 2012; Hatchett et al. 2013; Gillespie and Dinsmore 2014; Smith and Dwyer
2016). Wind farm implementation could lead to changes in predator abundance, which in turn
could explain some indirect impacts of wind farms on avian survival and breeding success
(Smith and Dwyer 2016). Indeed, habitat changes associated with other human-made structures,
such as natural gas infrastructures, have been previously discussed as potential drivers of bird
population declines through the increase in nest predation rates (Hethcoat and Chalfoun 2015a),
among other mechanisms.

Landscape transformation associated with wind farm development may alter predation-prey
interactions by increasing the abundance of potential predators, or favouring habitat utilization
by predators. The installation of linear corridors, such as access gravel-roads to wind turbines,
could act as movement pathways for potential mammalian predators (Trombulak and Frissell
2000; Frey and Conover 2006; Van Der Ree et al. 2015), generating shifts in the structure and
composition of communities (e.g., augmentation of generalist predators; Degregorio et al. 2014;
Hethcoat and Chalfoun 2015a). Additionally, collision events with wind turbines might attract
avian and mammalian predators (Rogers et al. 2014) because of the higher availability of food
resources in their vicinity, as has been described from roadkill carcasses (Van Der Ree et al.
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 km². 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...
with and without wind farms were of similar size (Mean ± SD; 135.06 ± 160.92 ha vs. 85.52 ± 59.52 ha in patches with and without wind farms, respectively; One-way ANOVA, F-value = 0.70, p = 0.42) and elevation (1164 ± 86 m vs. 1113 ± 58 m; One-way ANOVA, F-value = 1.75, p = 0.21).

2.2 Abundance of large mammalian and avian predators

In May 2016 and along one 500-m transect per plot, we counted all scat belonging to large mammals commonly described as nest predators in shrub-steppes (Pérez-Granados et al. 2017; Suárez et al. 1993; Yanes and Suarez 1996). Number of scats was used as a proxy of large mammalian predator abundance (Güthlin et al. 2014). Transects were conducted along the access path or gravel-road to the plot, and they were centred in order to be representative of the whole plot (Fig. 1). Transects were walked at constant speed and all scat found up to 1 m from both sides of the observer were recorded and identified to species level (Bang et al. 2007). Scats that could not be identified to species level were classified as unknown.

The abundance of avian predators was recorded from April to June 2016 using point-count stations. Point-count stations were distributed over the study period, performing at least one point-count station per month (April, May and June) in each plot. They were carried out by a single observer, and under similar favourable weather conditions (i.e., absence of precipitation and fog). Point-count stations (n = 188) were located both at the centre of each plot and randomly spread throughout the study area, at distances from 68.3 up to 894.7 m to the centre of the plot (mean ± SD = 475.2 ± 258.6 m). All potential avian nest predators (in accordance with Suárez et al. 2009) sighted in a 500 m buffer around the point-count station were recorded, but only those located within a 1 km radius around each plot were considered in subsequent analyses. In case of the slightest indication that two observations belonged to the same

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individual, one of them was discarded. Number of point-count stations in each plot ranged
between 9 and 40 (10.44 ± 3.31) and they extended between 20 and 410 mins (71.58 ± 79.07
mins), from sunrise to 2 p.m. Total effort time per plot ranged between 3.75 and 34.70 hours
(12.48 ± 11.20 h). Due to its random distribution, number of point-count stations located in the
surroundings of each plot and their duration differed between plots; therefore, the sampling effort
(i.e. number and duration of point-count stations) assigned to each plot was incorporated as an
offset in posterior analyses (see 2.5 Statistical analysis).

2.3 Nest predation experiment

We placed nine artificial nests at each plot. Artificial nests were placed in a cross-shape and at 50
metres interval (Fig. 1). Under this layout, plots at wind farm sites were comprised of artificial
nests located in a distance gradient to wind turbines (Fig. 1). Artificial nests were placed in the
same location in two sampling periods in 2016: at the beginning (April) and at the end (June) of
the breeding season. Artificial nests were monitored after 12 days, which corresponds to the
average incubation period of ground-nesting species inhabiting in the study area (e.g., 11 days
Eurasian skylark, 12 days Dupont’s lark, 14 days tawny pipit; Calero-Riestra et al. 2013; Pérez-
Granados et al. 2017; Suárez et al. 2009, 2005). Artificial nests were considered as predated
when at least one egg was either absent or damaged. Artificial nests damaged by ants were
discarded (n = 3) because they had holes that might attract predators or invalidate the artificial
nest. In addition, a whole plot (i.e., nine artificial nests) was discarded during the second
sampling period due to the placement of beehives in the surroundings, which may have affected
predator’s behaviour. Thus, a total of 161 (99 at control and 62 at wind farm sites) and 151 (89 at
control and 62 at wind farm sites) artificial ground-nests were monitored during the first and
second sampling period, respectively (total of 312 artificial nests).
Artificial nests were placed that resembled local natural nests. They were laid on the ground oriented towards N-NE (Yanes et al. 1996; Pérez-Granados et al. 2017) and next to a small shrub, covering ca. 50% of the surface area occupied by the artificial nest (see Online Appendix A). We tried to avoid differences in microhabitat variables in order to minimize their effect on predation rates, although they were incorporated in the analysis in order to control for their potential effect (see 2.4 Explanatory variables). Artificial nests were placed in a scrape dug into the ground with no lining, minimizing handling to avoid human odour that could attract predators. Artificial nests were baited with three quail (*Coturnix coturnix*) eggs. All eggs had the same origin, and they were handled under the same conditions (i.e. water-washed hands and rubber protective gloves). One of the three eggs was emptied, filled with plaster, and tied to a shrub with monofilament to avoid removal by predators and in order to potentially identify predator class (i.e., large/small mammalian or avian) using the cues left at the plaster. Highly disturbed artificial nests (i.e., the plaster egg was torn and detached from the monofilament) where the vegetation in the surroundings was matted down and/or the soil removed, were considered indicative of large mammalian or mesocarnivore predators. Remains of eggshell at the artificial nest and marks from pecks on the plaster egg were considered indicative of bird predation. Similarly, eggshell remains and small marks on the plaster egg clearly resembling the teeth of a small rodent were considered indicative of small mammal predation. In the absence of clear cues, potential nest predator was classified as unknown. Similar criteria for distinguishing these classes of nest predators have been employed before (Best 1978; Yanes and Suarez 1996).

2.4 Explanatory variables

We characterized artificial nest locations at two spatial scales: 1) landscape scale, referring to the habitat surrounding the artificial nest (i.e., measured in a 500-m radius-buffer around the centre
of each plot, hereafter plot level), and 2) microhabitat scale, referring to the immediate
surrounding of the artificial nest (i.e., measured at/from the artificial nest, hereafter nest level).

At the plot level, we considered wind farm occurrence, landscape features, and two indices of
predator abundance (Table 1). At the nest level, we incorporated the sampling period, the
proximity to sources of potential predators, and plant structure (Table 1).

Wind farm occurrence – Presence/absence of wind turbines in the shrub-steppe patch where plots
were located.

Sampling period – In order to control for potential seasonal patterns on nest predation rates
(Evans et al. 2005), we recorded the period at which the artificial nests were located (i.e., at the
beginning or at the end of the breeding season).

Landscape features – We characterized the landscape in a 500-m buffer around each plot (Fig.
1). We employed the Land Cover and Use Information System of Spain (SIOSE 2011) to
measure the surface area of the three major land-use types present in the study area: gravel-roads,
crops, and trees. These variables were calculated with the software QGIS 2.14.0 (Quantum GIS
Development Team 2019).

Proximity to sources of potential predators – We measured the distance from each artificial nest
to landscape structures that may be source of potential predators: distance to the nearest 1) tree;
2) crop; and 3) path or gravel-road. The distance to the nearest tree was estimated in the field
using a laser rangefinder (Leica 1200RF), whereas all the remaining variables were calculated in
QGIS 2.14.0 (Quantum GIS Development Team 2019).

Indices of predator abundance – Relative abundance of large mammalian (scat counts) and avian
predators per plot (see 2.2. Abundance of mammalian and avian predators).
Plant structure – We carried out one vegetation sampling at each artificial nest location to control for the potential effect of microhabitat on the probability of nest predation. Vegetation structure was measured in 1x1 m quadrats to record nest concealment, placing the artificial nest in the centre of the quadrat. We measured 13 vegetation structure variables describing cover (horizontal structure) and architecture (vertical structure) at each artificial nest (see Rotenberry and Wiens 1980 or Morales et al. 2008 for a similar approach). The importance of sampling both structures (horizontal and vertical) relies on the effect of cover and height on nest concealment and ultimately, on nest predation (Barrientos et al. 2009; Ponce et al. 2018). In each quadrat we measured the following variables related with vertical plant structure: 1) maximum height; and number of contacts at 2) 0-5 cm; 3) 5-10 cm; 4) 10-30 cm; and 5) above 30 cm height. Additionally, we measured several descriptors of horizontal structure: 1) total vegetation cover (%); 2) bare ground cover; 3) rock cover; 4) shrub cover; 5) perennial and annual herbaceous cover; 6) detritus cover; 7) lichen cover; and 8) moss cover.

2.5 Statistical analysis

Fixed covariates were z-standardized (i.e., mean 0 and standard deviation 1) and tested for collinearity prior to data analysis, retaining those predictors with a generalized variance inflation factor (GVIF) lower than 2 (Fox and Monette 1992).

2.5.1. Plant structure

Regarding the variables describing plant structure (Table 1), we carried out Principal Component Analysis (PCA) to reduce the number of predictors and to obtain interpretable gradients from an ecological point of view. Two different PCAs were carried out incorporating those variables related with: 1) vertical plant structure; and 2) horizontal plant structure. We retained those
components with an eigenvalue greater than 1. The function \textit{prcomp} of the package “stats” from the R software (v.3.6.2; R Core Team, 2019) was used.

2.5.2. \textit{Relationship between wind farm occurrence and gravel-roads}

In order to identify whether gravel-roads were related to wind farm occurrence, we fitted two linear models (Gaussian error distribution) to determine the best specification of random effects: 1) a spatial model, with a spatial random effect to control for potential spatial dependency (more details below); and 2) a non-spatial model, without random effects. In both models, we incorporated the surface area of gravel-roads as response variable and the factor ‘wind farm occurrence’ as predictor. Spatial and non-spatial models were compared using the Watanabe-Akaike information criterion (wAIC; Watanabe 2010), in which lower values means better fit. We only present the results for the best model (lowest wAIC value), and we selected the spatial model in case of similar wAIC values (i.e., ΔwAIC ≤ 2) because it produces more conservative results (Zuur et al. 2017).

The spatial model was fitted to account for potential non-independence of data collected from nearby plots. This was carried out by incorporating a spatial random effect using Integrated Nested Laplace Approximation with Stochastic Partial Differential Equations (INLA-SPDE; Lindgren et al. 2011). This method is an efficient and accurate way to model ecological data while accounting for spatial and temporal dependencies (e.g., Myer et al., 2017; Ward et al., 2015). Briefly, spatial dependency is accounted for using a latent Gaussian Markovian Random Field (GMRF) and the covariance matrix of the spatial residual term is approximated via SPDE framework. For that, we built a two-dimensional irregular grid (i.e., mesh) using a non-convex boundary for the coordinates of all plots, and with a buffer zone in order to avoid edge effects for the bordering vertices (Zuur et al. 2017) (see Online Appendix B). We used Penalized
Complexity priors (PC; Simpson et al. 2017) for the SPDE parameters to penalize the complexity of the GMRF and thus, control flexibility, reduce over-fitting, and improve predictive performance (Simpson et al. 2017; Fuglstad et al. 2019; Gómez-Rubio 2020). In accordance with the study spatial dimension and the data, we set an uninformative PC-prior for the range (i.e., the distance at which the spatial dependency diminish, $r$) so that $P(r < 7) = 0.5$, since 50% of the distances between plots were ca. 7 km, and a PC-prior for the standard deviation where $P(\sigma > 3) = 0.05$.

2.5.3. Predator abundance

Due to the positive relationship between wind farms occurrence and the surface area of gravel-roads (see 3. Results) we assessed the effect of both predictors on predator abundance independently. Thus, we carried out two independent analyses for each response variable (large mammalian and avian predator abundance): 1) addressing the effect of wind farm occurrence per se (factor ‘wind farms occurrence’ (presence/absence) as predictor); and 2) assessing the effect of landscape features that may affect habitat use by predators (landscape predictors measured at the plot level: surface area of gravel-roads, trees, and crops). In each set of analyses, we fitted a spatial and a non-spatial Poisson model (count data, log-link function). Both models were compared using wAIC to determine the best specification of random effects, and the best model was selected following the same procedure explained in 2.5.2. Relationship between wind farm occurrence and gravel-roads. In the case of avian predator abundance, we incorporated an offset to account for differences on sampling effort: the logarithm of the total time invested per plot. Overdispersion was assessed simulating 1,000 datasets from the model and calculating the dispersion statistic (Pearson residuals) for each simulated dataset $D|model$. Then, we compared the dispersion statistic of the observed data $D|data$ with the distribution of the dispersion.
statistic for the simulated data $D|model$. When $P(D|data > D|model)$ was either $\geq 0.975$ (overdispersion) or $\leq 0.025$ (underdispersion), the model was rejected.

2.5.4. Probability of nest predation

We split the set of predictors potentially affecting nest predation rates into three groups in order to avoid model overparameterization (Table 2). These groups were defined on the basis of two-fold criteria: 1) absence of collinearity among predictors; and 2) whether predictors were descriptors of the context or predator abundance, landscape, or microhabitat. For each set of predictors, we fitted a spatial and a non-spatial Bernoulli model (1 predated, 0 non-predated; log-link function), which were compared using wAIC values to select the best partial model (see 2.5.2. Relationship between wind farm occurrence and gravel-roads for further explanation).

Lastly, we fitted two Bernoulli models (spatial and non-spatial) incorporating only those predictors that showed an effect (i.e., 95% BCI for a given regression parameter did not contain zero) on the probability of nest predation in each partial model. Spatial and non-spatial Bernoulli models were compared by wAIC to select the best final Bernoulli model.

We fitted spatial models to account for non-independence of data collected from nearby artificial nests using the INLA-SPDE approach (same PC-priors for the SPDE parameters than in 2.5.2. Relationship between wind farm occurrence and gravel-roads). We built a two-dimensional irregular grid (i.e., mesh) using a non-convex boundary for the geographic coordinates of the artificial nests and with a buffer zone in order to avoid edge effects for the bordering vertices (Zuur et al. 2017) (see Online Appendix B). For the non-spatial model, we incorporated the artificial nest ID nested within plot as a random intercept to account for multiple observations from each artificial nest across plots (i.e., repeated-measures design, two sampling periods within...
the same year). All models described above were fitted using the R package “INLA” (Rue et al. 2009) in the free R software (v.3.6.2; R Core Team 2019). We used INLA default prior distributions for the intercept $\alpha \sim N(0,0)$, and the regression coefficients $\beta \sim N(0, 1000)$ (Gómez-Rubio 2020). Parameter estimates were reported as the posterior mean ($\beta$), associated standard deviation (SD) and the 95% Bayesian Credible Interval (95% BCI). We considered predictors to have an effect on the response variable when the parameter’s 95% BCI did not overlap zero (Zuur et al. 2017).

3. Results

3.1. Plant structure

The PCA on horizontal vegetation structure variables yielded four components (76.31% of total variance) explaining 28.41%, 18.94%, 15.90%, and 13.07% of variance, respectively (see Online Appendix C). The first component $PC1Hor$ represented a gradient on vegetation cover since it was positively associated with total vegetation cover (correlation coefficient = 0.85) and herbaceous cover (0.76), and negatively with bare ground cover (-0.75). The second component $PC2Hor$ was positively associated with shrub cover (0.89) and negatively with rock cover (-0.58). The third component $PC3Hor$ was negatively related with detritus (-0.68) and moss cover (-0.69), and the fourth component $PC4Hor$ defined a negative gradient of lichen cover (-0.72). On the other hand, the PCA on the vertical vegetation structure variables yielded two components (62.89% of total variance, see Online Appendix C) which explained 41.63% and 21.27% of variance, respectively. The first component $PC1Ver$ defined a positive gradient on the number of contacts below 30 cm: number of contacts at 0-5 cm (0.67), 5-10 cm (0.77), and 10-30...
Lastly, the second component $PC2Ver$ defined a negative gradient on maximum vegetation height (-0.78).

### 3.2. Relationship between wind farm occurrence and gravel-roads

The spatial model performed better ($wAIC = -106.7$) than the non-spatial model ($wAIC = 30.6$; Online Appendix D). The surface area of gravel-roads was higher at wind farm sites ($\beta \pm SD = 1.74 \pm; 95\% BCI = [1.19; 2.31]$; Fig. 2).

### 3.3. Predator abundance

Relative abundance of large mammalian predators (i.e., scat counts) ranged between zero and six per plot ($mean \pm SD = 1.22 \pm 2.02$). Scats belonged to red fox ($Vulpes vulpes; 68.2\%$), domestic or feral dog ($Canis lupus familiaris; 22.7\%$), and wild boar ($Sus scrofa; 9.1\%$). The number of avian predators recorded per hour ranged between zero and $0.74$ ($mean \pm SD = 0.18 \pm 0.22$). Potential avian predators registered were: carrion crow ($Corvus corone; 48.8\%$), common raven ($Corvus corax; 21.9\%$), Eurasian magpie ($Pica pica; 17.1\%$), and Montagu’s harrier ($Circus pygargus; 12.2\%$).

Comparison of $wAIC$ values indicated that spatial Poisson models performed better or similarly than non-spatial models (delta-$wAIC$ between 0.01 and 27.7; Online Appendix D), except for the model addressing the effect of landscape features on the relative abundance of large mammalian predators for which the non-spatial model was selected (non-spatial model $wAIC = 69.9$ and spatial model $wAIC = 6.1 \times 10^{17}$). However, parameter estimates were similar across both spatial and non-spatial models in all cases (see Online Appendix D). Poisson models did not show indications of overdispersion (Online Appendix E). The relative abundance of large mammalian predators was higher at wind farm sites ($\beta \pm SD = 2.33 \pm 1.34; 95\% BCI = [0.17;
5.51]) compared with control sites. However, the abundance of avian predators did not differ between control and wind farm sites ($\beta \pm SD = 0.09 \pm 0.50$, BCI = [-0.93; 1.07]). Regarding the effect of landscape features on predator abundance, the surface area of gravel-roads and trees had a positive effect on the relative abundance of large mammalian predators (Fig. 3; Table 3), whereas landscape variables did not influence the relative abundance of avian predators (Table 3).

3.4. Probability of nest predation

During the first survey at the beginning of the breeding season, a total of 22 out of 161 artificial nests (13.6%) were predated, whereas 34 out of 151 artificial nests (22.5%) were predated at the end of the breeding season (i.e., second sampling period). Thus, a total of 56 artificial nests were predated during this experiment, which results in an average predation rate of 17.9%. Twenty-seven out of the 56 artificial nests predated during the whole study period (48.2%) were located at wind farm sites and 29 predated artificial nests (51.8%) were located at control sites. Moreover, 38 artificial nests (67.9%) showed signs of large mammalian predation, five artificial nests (8.9%) of bird predation, and five artificial nests (8.9%) of small mammal predation. Eight artificial nests (14.3%) did not show clear cues and we were unable to identify predator species.

Comparison of wAIC values indicated that spatial Bernoulli models performed better or similarly than non-spatial models. Thus, we present results from spatial Bernoulli models although parameter estimates were similar across both models (see Online Appendix D). The final spatial Bernoulli model incorporating all important predictors from the partial models (Table 4) showed that the probability of nest predation differed between sampling periods, being higher at the end of the breeding season (Table 5). Additionally, the probability of nest predation increased when
the cover of moss and detritus was lower around the nest (i.e., positive effect of PC3Hor; Table 5), and when the surface area of gravel-roads and crops increased (Table 5; Fig. 4). Wind farm occurrence and the relative abundance of large mammalian and avian predators did not affect nest predation (Table 4).

4. Discussion

Our results suggest that wind farms might increase nest predation rates through the increase in relative abundance of large mammalian predators. Although wind farms did not directly increase nest predation rates, we did see higher relative abundance of large mammalian predators there. We also found that gravel-roads, which are more abundant on wind farms, were associated with increased relative abundance of large mammalian predators and increased nest predation rates. This connection points to a potential mechanism underlying the previously detected impact of wind farms on population trends of an endangered shrub-steppe species in the study area, the Dupont’s lark (Gómez-Catasús et al. 2018a), since displacement or changes in nest-site selection seems unlikely given the overall declining population trends experienced by the species (Gómez-Catasús et al. 2018b), including those populations in the absence of wind farms (Gómez-Catasús et al. 2018a). In this previous study, we observed that the Dupont’s lark experienced a 21% average annual decline in the presence of wind farms, ca. four times higher than in those populations in the absence of wind infrastructures (5.8% average annual decline; Gómez-Catasús et al. 2018a). The last finding of our research highlights the negative effect of crops interspersed in natural steppe landscapes on ground-nest predation rates. This result, together with the effect of gravel-roads, suggests that transformations of the shrub-steppe landscape may jeopardize 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
areas may offer shelter and food for mammalian predators, potentially explaining the higher relative abundance of these predators in the surroundings.

Wind farms may cause avoidance behaviour in upland bird species (Pearce-Higgins et al. 2009), and has been discussed as a potential driver of lower avian predator abundance or richness in the surroundings of these infrastructures (Garvin et al. 2011; Keehn and Feldman 2018; Thaker et al. 2018). In this study, the abundance of avian predators was not affected by either the presence of wind farms or by the landscape features in the surroundings. One possible explanation is that control and wind farm plots were located relatively close (mean ± SD = 3.28 ± 0.92 km) compared to home-range sizes of the avian species recorded, and hence both plots with and without windfarms may fall within the home range of a single individual. Moreover, the species forming the predator assemblage (corvids and harriers) in our study area may show complex and species-specific responses (Chalfoun et al. 2002; Tewksbury et al. 2006), hindering detection of a common effect on the avian predator community as a whole. The probability of collision and thus avoidance rates at wind turbines depends on myriad factors such as species-specific flight behaviour and morphology (Barrios and Rodríguez 2004; De Lucas et al. 2008). Some raptors and less manoeuvrable species have shown clear patterns of avoidance of wind turbines (De Lucas et al. 2008), but there are equivocal evidences about the effect of wind farms on the collision risk and avoidance behaviour of corvids and harriers (Devereux et al. 2008; Wilson et al. 2017; Bose et al. 2018), potential avian predators that co-occurred in our study area. Similarly, the absence of effect of landscape features could be because avian predators present in our study area differ in their habitat requirements or because the 500-m buffer that we used to characterize the landscape may not be big enough to capture the habitat used for avian predators. Therefore, local abundance of avian predators could not be explained through landscape features
or habitat changes induced by the implementation of wind turbines in shrub-steppes, and further research is needed to address predator-specific responses. In any case, birds have been described as uncommon nest predators in shrub-steppes (Calero-Riestra et al. 2013; Pérez-Granados et al. 2017; Yanes and Suarez 1996) and our results suggest that their density is low in the study area (mean ± SD = 0.18 ± 0.22 avian predators per hour).

We found that the probability of nest predation was higher at the end of the breeding season (June) as compared to the beginning of the breeding season (April). Our results are in agreement with previous studies that also reported an increase on the probability of nest predation later in the breeding season (Evans et al. 2005; Grant et al. 2005; but see Calero-Riestra et al. 2013). This pattern might be associated with an increase on predator activity over the season (Hatchett et al. 2013), due to a higher abundance of post-breeding adults and dispersal juveniles at the end of the breeding season. Indeed, potential trade-offs have been described in multi-brooded species, and specifically in ground-breeders, associated with early nesting at the beginning of the breeding season even though optimal conditions have not been reached at that moment (i.e., lower food availability), compared with later broods at the end of the breeding season when food availability is higher but also the risk of predation (Evans et al. 2005).

On the other hand, the probability of nest predation was not affected either by the occurrence or the proximity to wind turbines. Previous studies did not detect an effect of wind turbines on nest survival (Hatchett et al. 2013; Gillespie and Dinsmore 2014), while others detected an increase on nesting success closer to turbines probably due to a lower activity of raptors in the vicinity of wind farms (Rubenstahl et al. 2012). However, most of these studies did not incorporate measures of landscape changes associated to wind energy development, which may be better indices for quantifying the effects of wind energy development on nest predation rates. For
instance, Hethcoat and Chalfoun (2015b) found that in natural gas development facilities, well density resulted in overly conservative estimates of nest survival probability, whereas predictors quantifying landscape transformation (i.e., sagebrush habitat loss) negatively influenced nest survival rates. Similarly, Mahoney and Chalfoun (2016) reported an increase on nesting success as turbine density decreased. In accordance with these results, we did not detect a direct effect of wind turbine occurrence on the probability of ground-nest predation, but we did find evidence for a positive effect of the amount of gravel-roads in the surroundings, a landscape transformation associated with wind farm implementation (Fig. 2). Indeed, the probability of nest predation drastically increased above 2 ha of gravel-roads (i.e., 2.5% of the surface area covered by the 500 m buffer around each plot at which the landscape was characterized; Fig. 4) and a 25% predation probability was reached at 4 ha (i.e., 5% of the total surface area; Fig. 4), suggesting that small proportions of this landscape feature can compromise the breeding success of ground-nest breeders.

Consistent with our predictions, the relative abundance of large mammalian predators and the probability of nest predation were higher in shrub-steppe patches with a higher surface area of gravel-roads. However, no direct effect of predator abundance indices, either avian or large mammalian, on the probability of nest predation was detected. One possible explanation is that the samplings of the relative abundance of large mammalian (late May) and avian predators (from April to June) are decoupled from each specific nest predation experiment (April and June, respectively), hindering to detect an effect on the probability of nest predation. Moreover, nest predation could depend on the activity of a particular predator species and not on the whole predator assemblage (Benson et al. 2010; Hethcoat and Chalfoun 2015a). Lastly, one pitfall of our study design is that abundance of other potential predators, also linked to roads, was not
properly surveyed. This is the case for small mammals or reptiles, which have been recorded as potential predators of our artificial nests (8.9% of the artificial nests for small mammals) and described as common nest predators in shrub-steppes (garden dormouse *Elyomis quercinus*, ocellated lizard *Timon Lepidus*, and lataste viper *Vipera latastei*; Pérez-Granados et al. 2017; Suárez et al. 1993; Yanes and Suarez 1996). Roadside verges have been described as refuges for small mammals (Ruiz-Capillas et al. 2013) and are used by reptiles for thermoregulation (Ashley and Robinson 1996). Indeed, other studies have reported cascade impacts of wind farms on the density of ground-dwelling prey (e.g., lizards) as a consequence of the lower abundance and activity of predatory birds (Thaker et al. 2018). Therefore, future studies should address the effect of wind farms on this predator community and its role on ground-nest predation.

The expansion of crops as a result of agricultural intensification is one of the major sources of habitat loss and fragmentation on shrub-steppe ecosystems (Laiolo and Tella 2006) that might affect nest predation in several non-mutually ways (e.g., increasing predator abundance or nest densities in suboptimal habitats; Evans, 2004). In this study, we found a positive association between crop surface area and the probability of nest predation. The role of non-cropped habitats (e.g. field margins) as undisturbed and stable refuges for mammals and other potential predators is well-known (Benton et al. 2003; Rodríguez-Pastor et al. 2016) thus increasing predation risk on nests located closer to agricultural edges (Ponce et al. 2018). Similarly, stone walls used to delimit crop fields in our study area are commonly used by lacertids and other reptiles (Díaz et al. 2006). Therefore, it is reasonable that the probability of nest predation increases with the surface area of crops, which might be an important source of potential predators. In particular, the probability of nest predation was null or low when the surface area of crops in the surroundings was lower than 20 ha (i.e., ca. 25% of the total surface area characterised in the
surroundings within the 500 m buffer; Fig. 4), and it greatly increased when the surface area of crops was above 20 ha, reaching ca. a 50% predation probability when the surface area of crops was above 40 ha (i.e., ca. 50% of the total surface area; Fig. 4). Thus, the effect of crops on predation rates might be specially worrying if crop amount is above 25% of the total surface area in the surroundings. Lastly, microhabitat variables such as vegetation cover and height, also influence nest predation rates (Barrientos et al. 2009; Praus and Weidinger 2015; Ponce et al. 2018). In this study, the probability of nest predation decreased with the cover of moss and detritus, whereas the remaining microhabitat variables did not determine nest predation rates. We attempted to minimize the differences on microhabitat variables when placing the artificial nests (i.e., under a short shrub covering ca. 50% of the nest). Therefore, it is reasonable that most of the microhabitat variables related with plant structure did not influence nest predation rates. One possible explanation for the effect of moss and lichen cover on predation rates is that they may offer better nest concealment and reduce nest conspicuousness, thus reducing nest predation (Hatchwell et al. 1996).

The use of artificial nests to answer ecological questions is under debate (see review in Moore and Robinson 2004). The degree to which results from artificial nests can be generalized to real bird populations has been questioned because artificial nests might attract different predators than real nests and hence, may be predated at different rates (Thompson and Burhans 2004). Nest predation rates in our study (17.9%) were similar than predation rates previously described in our study area from natural nests during the incubation period (18.9%, n = 27; Pérez-Granados et al. 2017). Moreover, the predation rate obtained after monitoring 27 nests of black-eared wheatear (3), Dupont’s lark (7), greater short-toed lark (6), Eurasian skylark (10) and tawny pipit (1) during the incubation period in the study area (years 2017-2019) was 33.3% (authors’
unpublished data). In any case, neither breeding success nor overall predation rates were within the aims of this study. A drawback of our study is that nest predators might not be reliably assigned based on cues left at artificial nests (Williams and Wood 2002; Thompson and Burhans 2004) and thus, these results may be misinterpreting the importance of some predators (Benson et al. 2010). Our goal was to identify main nest predators in order to contextualize factors at microhabitat and landscape scales determining the probability of nest predation. We consider that we succeeded in this purpose since mammals have previously been described as main predators from natural nests in shrub-steppes, whereas reptiles, birds and small mammals were considered as rare (Suárez et al. 1993). Therefore, while our results cannot be used to obtain predation rates at natural nests, we consider that they provide important insights on spatial trends in predation that may have implications for natural nests. Moreover, some authors admit that they provide useful data with larger sample sizes as compared to those from natural nests due to the time-consuming task of finding real nests, highlighting their utility for comparative purposes (Major and Kendal 1996; Evans 2003) and as indices of relative predation pressure (Vögeli et al. 2011). Thus, main conclusions drawn from this study about the role of gravel-roads and crops as sources of predators remain reliable.

5. Conclusions and implications for conservation

This paper delves into a potential mechanism explaining the negative effects of wind farms on ground-nesting birds: the increase of large mammalian predator abundance through landscape transformation that may lead to higher ground-nest predation rates. Specifically, this indirect impact has been addressed in shrub-steppes, a key ecosystem for the preservation of avian wildlife since shrub-steppe birds experience declining population trends (Burfield 2005;
Sanderson et al. 2005), which might be aggravated in the presence of wind infrastructures (Gómez-Catasús et al. 2018a). Future research should address the effects of wind farms and landscape features on avian breeding success monitoring real nests throughout all stages of the nesting cycle, as well as studying patterns of nest-site selection or the effect of turbine noise on predator-prey interactions.

Predator control has been widely proposed as an effective conservation measure of vulnerable bird populations (Smith et al. 2010). However, a deeper understanding of the mechanisms that relate wind farm occurrence, habitat, and landscape to breeding success and other predators not specifically covered by this study (e.g., small mammals and reptiles) is needed to determine whether these management practices could be effective in shrub-steppe ecosystems. Meanwhile, managers should prioritize alternative solutions such as installing narrow and straight access-roads or reusing already-existing roads, along with placing wind turbines outside the distribution area of endangered bird species. These specific recommendations aim to minimize the development of gravel-roads for access and maintenance of wind turbines during the design of future wind energy projects, in order to avoid shifts on the predator community and the potential increase on predation rates.

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Electronic supplementary material

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

Declarations
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Conflicts of interest/Competing interests
None

Ethics approval (include appropriate approvals or waivers)
Not applicable

Consent to participate (include appropriate statements)
Not applicable

Consent for publication (include appropriate statements)
Not applicable

Availability of data and material (data transparency)
Data will be submitted to an appropriate public data repository

Code availability (software application or custom code)
R-code will be submitted to an appropriate public repository

Author contributions

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https://doi.org/10.2307/1936840

https://doi.org/10.1894/0038-4909-57.2.189


https://doi.org/10.1016/j.biocon.2012.09.025


Table 1. Explanatory variables incorporated in the analyses carried out in this study. The level (nest or plot) at which the different variables were measured is indicated. In addition, the analysis in which the explanatory variables were incorporated as predictors is stated. PA: Predator abundance. NP: Nest predation

<table>
<thead>
<tr>
<th>Environmental factor</th>
<th>Variables</th>
<th>Level</th>
<th>Analysis</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wind farm occurrence</td>
<td>Presence/Absence of wind turbines</td>
<td>Plot</td>
<td>PA, NP</td>
</tr>
<tr>
<td>Sampling period</td>
<td>Beginning or end of the breeding season</td>
<td>Nest</td>
<td>NP</td>
</tr>
<tr>
<td>Landscape features</td>
<td>Surface area of gravel-roads (m²)</td>
<td>Plot - 500 m buffer</td>
<td>PA, NP</td>
</tr>
<tr>
<td></td>
<td>Surface area of crops (m²)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Surface area of trees (m²)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Proximity to sources of potential predators</td>
<td>Distance to the nearest path/gravel-road (m)</td>
<td>Nest</td>
<td>NP</td>
</tr>
<tr>
<td></td>
<td>Distance to the nearest crop (m)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Distance to the nearest tree (m)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Indices of predator abundance</td>
<td>Large mammalian predators</td>
<td>Plot – 500 m transect</td>
<td>NP</td>
</tr>
<tr>
<td></td>
<td>Avian predators</td>
<td>(large mammalian predators)</td>
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</tr>
<tr>
<td></td>
<td></td>
<td>and 1 km buffer (avian</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>predators)</td>
<td></td>
</tr>
<tr>
<td>Plant structure</td>
<td><strong>Vertical structure:</strong></td>
<td>Nest - 1x1 m quadrat</td>
<td>NP</td>
</tr>
<tr>
<td></td>
<td>Maximum height (cm)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Number of contacts at 0-5cm, 5-10cm, 10-30cm and &gt;30 cm</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td><strong>Horizontal structure:</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Total vegetation cover (%)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Bare ground cover (%)</td>
<td></td>
<td></td>
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<tr>
<td></td>
<td>Rock cover (%)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Shrub cover (%)</td>
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<td></td>
</tr>
<tr>
<td></td>
<td>Herbaceous cover (%)</td>
<td></td>
<td></td>
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<tr>
<td></td>
<td>Detritus cover (%)</td>
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<tr>
<td></td>
<td>Lichen cover (%)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Moss cover (%)</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Table 2. Partial Bernoulli models fitted to address the factors explaining the probability of nest predation at 312 artificial nests placed in two occasions (April and June) in 2016. Predictors incorporated in each partial model are indicated.

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

\(^a\)PC1Hor-PC4Hor states for the Principal Components yielded by the PCA on the horizontal plant structure, whereas PC1Ver-PC2Ver are the Principal Components yielded by the PCA on the vertical plant structure (see Results).

Table 3. Results of the non-spatial Poisson model assessing the effect of landscape features on the relative abundance of large mammalian predators and the spatial Poisson model assessing the effect of landscape features on the relative abundance of avian predators (n=18 plots in 2016). Posterior mean (\(\beta\)), standard deviation (SD) and 95% Bayesian Credible intervals (95% BCI) are shown for the intercept and each landscape variable. Predictors with an important effect on the relative abundance of large mammalian predators are highlighted in bold.

<table>
<thead>
<tr>
<th></th>
<th>Mammalian predators</th>
<th>Avian predators</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>(\beta)</td>
<td>SD</td>
</tr>
<tr>
<td>Intercept</td>
<td>-0.25</td>
<td>0.29</td>
</tr>
<tr>
<td>Surface area of roads</td>
<td><strong>0.63</strong></td>
<td>0.18</td>
</tr>
<tr>
<td>Surface area of trees</td>
<td><strong>0.52</strong></td>
<td>0.23</td>
</tr>
<tr>
<td>Surface area of crops</td>
<td>-0.40</td>
<td>0.35</td>
</tr>
</tbody>
</table>
Table 4. Results of the three partial spatial-Bernoulli models addressing the factors explaining the probability of nest predation at 312 artificial nests placed in two occasions (April and June) in 2016. The posterior mean (β), standard deviation (SD), and 95% Bayesian Credible Intervals (95% BCI) are shown for the intercept and each variable independently for each partial model. Predictors with an important effect on the probability of nest predation are highlighted in bold.

<table>
<thead>
<tr>
<th>Model</th>
<th>β</th>
<th>SD</th>
<th>95% BCI</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Context</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
<td>-3.43</td>
<td>1.01</td>
<td>[-5.66; -1.64]</td>
</tr>
<tr>
<td>Wind farm occurrence (Presence)</td>
<td>1.41</td>
<td>1.39</td>
<td>[-1.19; 4.37]</td>
</tr>
<tr>
<td><strong>Period (End breeding season)</strong></td>
<td><strong>0.73</strong></td>
<td><strong>0.34</strong></td>
<td><strong>[0.07; 1.41]</strong></td>
</tr>
<tr>
<td>Abundance of avian predators</td>
<td>0.55</td>
<td>0.54</td>
<td>[-0.44; 1.71]</td>
</tr>
<tr>
<td>Abundance of mammalian predators</td>
<td>-0.33</td>
<td>0.62</td>
<td>[-1.63; 0.87]</td>
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<td><strong>Landscape</strong></td>
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<td></td>
<td></td>
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<tr>
<td>Intercept</td>
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<td>[-4.30; -1.32]</td>
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<tr>
<td><strong>Surface area of gravel-roads</strong></td>
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<td><strong>0.46</strong></td>
<td><strong>[0.17; 2.01]</strong></td>
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<tr>
<td><strong>Surface area of crops</strong></td>
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<td><strong>0.48</strong></td>
<td><strong>[0.48; 2.37]</strong></td>
</tr>
<tr>
<td>Surface area of trees</td>
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<td>0.50</td>
<td>[-1.97; 0.01]</td>
</tr>
<tr>
<td>Distance to nearest path/gravel-road</td>
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<td>0.23</td>
<td>[-0.33; 0.56]</td>
</tr>
<tr>
<td>Distance to nearest tree</td>
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<td>0.24</td>
<td>[-0.65; 0.32]</td>
</tr>
<tr>
<td>Distance to nearest crop</td>
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<td>0.34</td>
<td>[-0.35; 0.98]</td>
</tr>
<tr>
<td><strong>Microhabitat</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intercept</td>
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<td>0.63</td>
<td>[-3.92; -1.43]</td>
</tr>
<tr>
<td>PC1Hor</td>
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<td>[-0.01; 0.92]</td>
</tr>
<tr>
<td>PC2Hor</td>
<td>0.32</td>
<td>0.23</td>
<td>[-0.11; 0.78]</td>
</tr>
<tr>
<td><strong>PC3Hor</strong></td>
<td><strong>0.73</strong></td>
<td><strong>0.27</strong></td>
<td><strong>[0.22; 1.30]</strong></td>
</tr>
<tr>
<td>PC4Hor</td>
<td>0.31</td>
<td>0.20</td>
<td>[-0.07; 0.71]</td>
</tr>
<tr>
<td>PC1Ver</td>
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<td>[-0.29; 0.45]</td>
</tr>
<tr>
<td>PC2Ver</td>
<td>0.16</td>
<td>0.20</td>
<td>[-0.22; 0.55]</td>
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</table>
Table 5. Results of the final spatial-Bernoulli model addressing the factors explaining the probability of nest predation at 312 artificial nests placed in two occasions (April and June) in 2016. The Posterior mean ($\beta$), standard deviation (SD) and 95% Bayesian Credible intervals (95% BCI) are shown for the intercept and each variable. Predictors with an effect on the probability of nest predation are highlighted in bold.

<table>
<thead>
<tr>
<th>Predictor</th>
<th>$\beta$</th>
<th>SD</th>
<th>95% BCI</th>
</tr>
</thead>
<tbody>
<tr>
<td>Intercept</td>
<td>-2.74</td>
<td>0.62</td>
<td>[-4.09; -1.64]</td>
</tr>
<tr>
<td>Period (End breeding season)</td>
<td>0.73</td>
<td>0.34</td>
<td>[0.07; 1.42]</td>
</tr>
<tr>
<td>Surface area of gravel-roads</td>
<td>0.76</td>
<td>0.39</td>
<td>[0.04; 1.60]</td>
</tr>
<tr>
<td>Surface area of crops</td>
<td>0.94</td>
<td>0.41</td>
<td>[0.20; 1.83]</td>
</tr>
<tr>
<td>PC3hor$^a$</td>
<td>0.56</td>
<td>0.24</td>
<td>[0.09; 1.05]</td>
</tr>
</tbody>
</table>

$^a$ Third Principal Component yielded by the PCA on horizontal vegetation structure variables. It was negatively related with detritus and moss cover.
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.
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.
Fig. 3 Relative abundance of large mammalian predators (i.e., scat counts) in relation to: (a) the surface area of gravel-roads (ha); and (b) the surface area of trees (ha) in a 500 metres (radius) buffer around each plot. Observed values for the plots located at control (n = 11; filled dots) and wind farm sites (n = 7; empty dots) are depicted. Mean (black line) and 95% Bayesian Credible Interval (grey surface) are represented.
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.