

MODEL-ASSISTED MONITORING OF BIODIVERSITY

Evaluating the regional cumulative impact of wind farms on birds: how can spatially explicit dynamic modelling improve impact assessments and monitoring?

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Summary

1. The Eurasian skylark *Alauda arvensis* is very susceptible to the negative effects of wind farms. In northern Portugal, this evidence is particularly severe due to the skylark's preference for mountain breeding habitats where most wind farms are located. Facing the frequent failure of environmental impact assessments (EIA) to evaluate the cumulative impacts of wind farms on wildlife, this study aimed to develop and test a methodology to quantify local and regional consequences on birds, using skylarks as a test species, taking into account future predictable environmental changes.

2. We propose a spatially explicit dynamic approach that combines the results from multiple modelling techniques under a common framework to assess the local and cumulative regional impacts of wind farms on skylark populations. This includes the following: (i) modelling the local impact of wind farms (in terms of collision mortality) on the skylark population dynamics by developing an index for quantitative assessments, (ii) determining the actual and future skylark breeding distribution across northern Portugal and (iii) integrating the above contributions in an emergent spatially explicit regional representation to capture the ecological cumulative consequences as a whole.

3. The simulations show an increasing average local impact for the skylark breeding populations directly affected by wind farms, expressed in mean number of collision fatalities per UTM study unit (1 km²), representing 1.3% of the local breeders in 2006 and 4% in 2026.

4. The distribution area of skylark breeding populations was predicted to decrease around 4.5% throughout a period of 15 years, as a result of the scenario of climate and land cover changes in the study area. When combined with a concomitant increase in skylark global mortality (c. 184%) induced by all wind farms in the study region, the above trend contributes to an intensification of the regional cumulative impact from 1.2% to 3.7% of the total estimated breeding individuals.

5. *Synthesis and applications.* The proposed modelling framework represents a step forward in evaluating the multi-scale cumulative consequences of wind farms on vulnerable birds, using skylarks as a test species. This could be used in the future to guide monitoring efforts and to improve the applicability of the data bases generated by long-term ecological research and monitoring studies.

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Introduction

Wind power generation has been the fastest growing source of renewable energy world-wide, which is leading to a rapid increase in the number of proposed new wind farms (Kaldellis & Zafirakis 2011). In the last two decades, the wind energy sector in Portugal had increased by around 41% in 2013, considering the total power produced from renewable energies at the national level (DGEG 2013). The total installed capacity reached 4452 MW, generated by 224 onshore wind farms placed throughout the mainland territory (DGEG 2013), from which around 42% of the total wind turbines (~1060 wind turbines) are located in the north of the country (i.e. NUTS2-PT11, considering the European Union nomenclature of territorial units for statistics; EUROSTAT 2014). The region is dominated by mountainous areas with the greatest wind potential and favourable remote locations for the placement of wind farms. Nevertheless, despite the clear benefits of wind energy, that is the reduction in greenhouse gas emissions (Panwar, Kaushik & Kothari 2011), these areas are also important for wildlife conservation, for example with several special protected areas and natural parks. In this context, the balance of cost–benefits, including the direct and indirect local impacts of wind farms on wildlife and nature conservation, that is the collision mortality of birds and bats, has not been met (Drewitt & Langston 2006). Environmental impact assessment (EIA) and general ecological monitoring (GEM) programmes may provide mechanisms to support sustainable development and the conservation of biological diversity (Söderman 2006). The universal application of EIA and GEM is the basis for implementing national strategies of sustainable development. EIA is used to foresee the environmental consequences of proposed human actions, and GEM is used to implement corrections and adaptations to reduce ecological impacts caused by humans (MacNeely 1994). Although the ecological inputs to environmental statements are considered in EIA, many criticisms have arisen from lack of scientific rigour and failure to evaluate the effective ecological impacts (Madsen *et al.* 2010).

The Eurasian skylark *Alauda arvensis* (Linnaeus, 1758), although a common and widespread breeding bird across Europe, is classified with an unfavourable conservation status (BirdLife International 2004) due to a severe decline associated with extreme habitat changes induced by new agricultural paradigms since the 1970s (Chamberlain & Crick 1999). Considering that the population densities remain far below the reference level that preceded this decline, the species is currently classified as ‘depleted’ at the European level (BirdLife International 2004). In

Portugal, although classified as ‘least concern’ (Cabrál *et al.* 2006), breeding skylark populations are almost exclusively distributed in the northern areas (Equipa Atlas 2008), selecting open mountain habitats that are heavily pressured by the increasing installation of wind farms. In this context, the skylark’s typical song flight, associated with mate choice and territoriality (Hedenström 1995), involving vertical flights that usually exceed 50 m in height, makes the species highly vulnerable to collision with wind turbines (Bernardino *et al.* 2010). Therefore, skylark populations from northern Portugal seem to be particularly susceptible to this anthropogenic disturbance, where the potential implications of wind farms at a regional scale may produce much larger effects (e.g. cumulative consequences of persistent collision fatalities) among other adverse global changes (e.g. induced by the ongoing climatic alterations). The combination of local population dynamics (temporal) and regional distributional (spatial) models can help to predict how anthropogenic and environmental changes will affect the abundance and displacement of vulnerable species or communities in disturbed ecosystems and regions (Guisan & Thuiller 2005). Dynamic models allow prediction of trends in the local population dynamics (e.g. survival rates, breeding productivity, reproductive success), including those attributable to changes in habitat due to different sources of perturbation (Topping, Odderskær & Kahlert 2013). Complementarily, spatial models are useful to identify areas where the conflict between the previous forecasted trends and drivers of pressure are of major conservationist concern, by allowing the integration of the ecological consequences from local to regional levels (Bjørnstad, Ims & Lambin 1999; Pearce-Higgins *et al.* 2008; Roscioni *et al.* 2013, 2014; Santos *et al.* 2013).

Combining multiple modelling techniques under a spatially explicit predictive framework could guide the assessment and timely mitigation of cumulative impacts while at the same time improving the strategic monitoring of wildlife change driven by specific processes (Bastos *et al.* 2012). In this context, this study aimed to develop and test a model-based methodology to predict and quantify both the local effects of each wind farm and the cumulative impacts of these infrastructures at regional level on birds, using the skylark’s breeding populations from northern Portugal as a test species. We propose a spatially explicit dynamic modelling approach that combines the results from downscaling techniques, species distribution models (SDMs), system dynamics (SD) and geographic information systems (GIS) under a common framework, supported by existing data bases. The framework proposal includes the following: (i) modelling the local impact of wind farms (in terms of collision mortality) and habitat

suitability on the skylark population dynamics by developing an index for quantitative assessments; (ii) determining the actual skylark breeding distribution area across northern Portugal and project a possible future scenario of alteration of the species distribution area according to expected climate and land cover changes; and (iii) integrating the above contributions in an emergent spatially explicit regional representation in order to capture the ecological cumulative consequences as a whole. These procedures were tested as a contribution to improve the applicability of the existing data bases and the quality of infrastructural impact evaluation produced in the scope of long-term impact assessments and/or monitoring programmes. Moreover, we discuss the implications of this new methodology for the improvement of monitoring practice, especially in the case of project-based impact assessments but also as part of wider, strategic biodiversity observation programmes.

Materials and methods

STUDY AREA

The study area is located in northern Portugal (Fig. 1), covering an area of 21 515 km², which includes the main skylark breeding distribution at the national level, and populations are mostly found at higher altitudes in mountainous areas (Equipa Atlas 2008; Catry *et al.* 2010). Although in the same area as several sites of the Natura 2000 network, the skylark distribution study area is highly pressured by the presence of wind farms (Fig. 1). This region is located in the transition of the Euro-Siberian (Atlantic) and the Mediterranean biogeographic regions (Costa *et al.* 1998), covering diverse climatic conditions due to its geographic position and rough topography (altitude range: 0–1545 m). The west part of the study area is characterized by a rainy and cold winter with a moderately hot and dry summer, particularly in low altitudes, while in the east the climate is

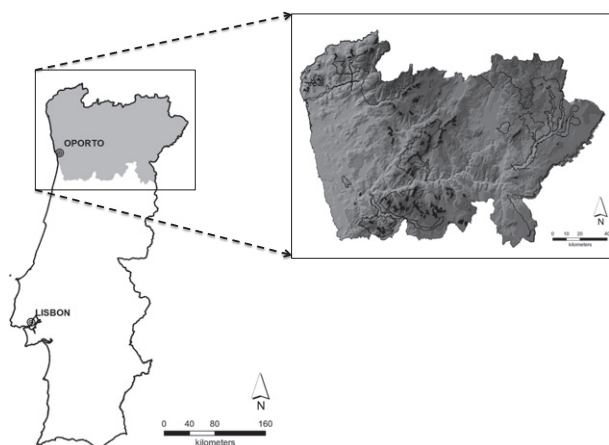


Fig. 1. Location of the study area in northern Portugal, the Natura 2000 network sites delimitation (black lines) and the wind turbine positions in the mountain areas throughout the region (black spots).

increasingly drier. Mean annual rainfall ranges from c. 400 mm in the eastern valleys to over 2500 mm in the western mountain summits. The marked regional heterogeneity of the environmental conditions, land uses, geology and topography supports a high diversity of vegetation cover throughout the study area (Caetano, Nunes & Nunes 2009).

THE MODELLING FRAMEWORK

The proposed modelling framework (Fig. 2) combines downscaling techniques, species distribution models, system dynamics and integrative spatial projections (geographic information systems), in order to assess the cumulative impacts of wind farms on skylark breeding populations in northern Portugal. The framework is initiated by the quantitative assessment of the local impact of wind turbines on the skylark breeding populations, designated by Local Impact Index. This index is calculated in terms of the skylark's collision mortality, taking into account the influence of local habitat dynamics and the effects caused by the presence of the wind turbines. In parallel, the skylark breeding distribution is downscaled from coarse-scale existing data and projected under credible scenarios of climate and land cover changes. These simulations, when superimposed and integrated with fine-grained predictions, will allow assessing, in space and time, the magnitude of the predicted local effects at the regional level. In fact, since the cumulative ecological consequences from the contributions of singular wind farms are synergetic (Masden *et al.* 2010), the combination of such modelling techniques is considered a promising approach to address complex emergent problems related to long-term impact assessments of wind farms on birds, from local to regional contexts.

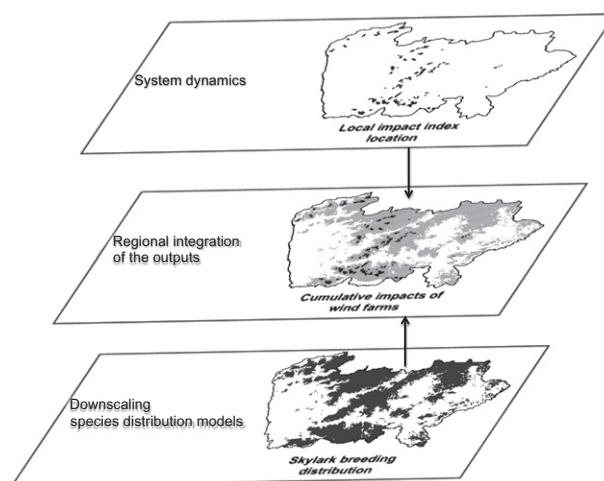


Fig. 2. Spatially explicit dynamic modelling approach to assess the cumulative impacts of wind farms on skylark populations from northern Portugal. This framework combines the dynamic modelling of the local impacts induced by wind farms (in terms of collision mortality) and habitat suitability on skylark breeding populations (Local Impact Index), the determination of the actual and future skylark breeding distribution considering expected climate and land cover changes across northern Portugal, and the integration of the above contributions in an emergent spatially explicit regional representation of the ecological cumulative consequences.

DYNAMIC MODELLING

Data resolution and study unit characterization

To model the skylark population dynamics, an UTM grid of 1-km² cells was superimposed onto the study area in order to characterize wind farm and land cover type variables. Wind farm location data were supplied by the Portuguese National Institute for Nature Conservation and Forests (ICNF) and overlaid to the UTM grid to identify the spatial units (i.e. 1-km² UTM cells) where wind turbines are or will be installed (i.e. study units). This group of study units was considered as the spatial universe for skylark population dynamics modelling under the direct influence of wind turbines. The total number of wind turbines present in each study unit and the timing of the respective installation were considered taking into account the known commissioning year of the wind farms (Wind Power 2013). Therefore, from the 22 013 UTM cells of the study area, 500 were selected as study units considering the presence or future installation of wind turbines (ranging from 1 to 10 wind turbines per study unit) from a total of 1192 wind turbines expected for the study region. Although most of the 123 wind farms were already installed in the study area between 1996 and 2011, 9 new wind farms were planned to be operational in 2014, representing 184 additional wind turbines. The Corine Land Cover 2006 (EEA, 2012) for the characterization of the land cover in the selected study units was used. Eight land cover categories were selected as representative of the land cover mosaic in which skylark breeding habitats are also present, as well as areas susceptible to becoming suitable habitats for the species, because of typical habitat dynamics in the region, mostly determined by fire events (Table 1).

Conceptualization of the population dynamics model

To predict the local combined effects of wind farms and habitat suitability on the skylark population dynamics (Fig. 3), four sub-models were designed in order to: (a) simulate the skylark population dynamics based on their reproductive biology; (b) reproduce the habitat dynamics taking into account the typical occurrence of fire events; (c) generate stochastic fatalities attributable to collisions with wind turbines; and (d) develop an index for the quantitative assessment of the local impact of wind farms in skylark population dynamics.

Table 1. Characterization of the land cover present in each study unit (1-km² UTM cell), considered for the skylark population dynamics modelling, with the respective Corine Land Cover (CLC) code, land cover description and the selected habitat classes used for the dynamic model construction

CLC code	Description	Model habitat classes
311	Broad-leaved forest	<i>Shrublands and forests</i>
312	Coniferous forest	<i>Shrublands and forests</i>
313	Mixed forest	<i>Shrublands and forests</i>
321	Natural grasslands	<i>Open lands</i>
322	Moors and heathland	<i>Shrublands and forests</i>
324	Transitional woodland shrub	<i>Shrublands and forests</i>
333	Sparsely vegetated areas	<i>Open lands</i>
334	Burnt areas	<i>Burnt areas</i>

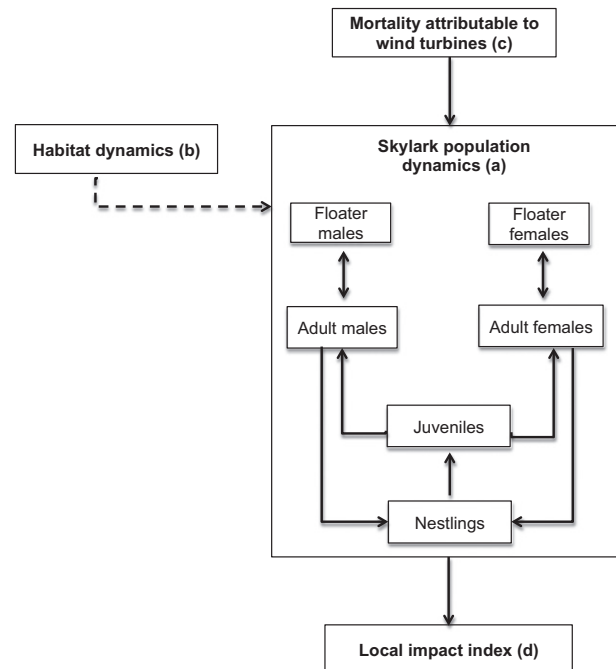


Fig. 3. Conceptual diagram of the dynamic model used to predict the local combined effects of wind farms and habitat suitability on skylark population dynamics in northern Portugal. The model is composed of different dynamic submodels and their interactions at the study unit level: (a) skylark population dynamics based on species reproductive biology, (b) the habitat dynamics taking into account the typical occurrence of fire events, (c) the influence of stochastic fatalities attributable to collisions with wind turbines and (d) the quantitative estimation of the local impact of wind farms (in terms of collision mortality) for skylark population dynamics (Local Impact Index).

Skylark population dynamics were expressed in densities (number of individuals per hectare). The time unit chosen was the month for a simulation period of 20 years, considered appropriate to capture population dynamics seasonality and to assess long-term impacts taking into account the average lifetime of wind farms. In order to reproduce complete breeding cycles from skylark nest-building to the emancipation of the young birds, three stages of the species' life cycle were considered: nestlings, juveniles and breeding adults (males and females; Fig. 3a; section 2 of Appendix S1, Supporting Information). The processes and factors intrinsically associated with skylark population dynamics were based on existing parameters from several studies concerning the species general demographic and phenological attributes, such as breeding period, clutch sizes, hatching success, nestling mortality, juvenile mortality from independence to age of first breeding, and adult survival rates (Table S1). Overall, the density of nestlings was generated taking into account the number of breeding pairs present in each study unit (Fig. 3a), the respective multi-brood attempts with realistic random clutch sizes (Delius 1965), and the natural mortality of nestlings (Shurulinkov 2005; section 2.1. of Appendix S1). Since resource availability limits the number of viable breeding territories, and competition makes some individuals leave when all suitable breeding territories are occupied (Delius 1965), we assumed density-dependent dispersal mechanisms to model the species population dynamics (section 2.3. of Appendix S1). In this context, floaters were assumed as

the individuals who re-enter into the reproductive population when a breeding territory or potential mate becomes available (Penteriani, Ferrer & Delgado 2011; Fig. 3a; section 2.4. of Appendix S1). For a better integration and comprehension of the abundances simulated for each study unit, the skylark density was automatically converted into the total number of individuals (section 3 of Appendix S2), considering the available area of favourable habitats, expressed dynamically by the combination of the three main classes of land cover selected: 'open lands', 'burnt areas' and 'shrublands and forests' (Table 1 and Fig. 3b; section 2 of Appendix S2). The class 'open lands', which includes natural grasslands and sparsely vegetated areas (Table 1), was assumed as the optimal breeding habitats for the species, since the skylark breeding distribution in the Iberian Peninsula is highly associated with shrub steppes (Suárez, Garza & Morales 2003). The class 'burnt areas' (Table 1) represents areas affected by fires within the study region, a stochastic but recurrent event with a typical influence on the local ecological successions (Moreno, Vazquez & Velez 1998). This class was also considered a suitable breeding habitat since the species prefer areas with low vegetation height and sparse ground covers for nesting (Toepfer & Stubbe 2001; Catry *et al.* 2010). Conversely, the class 'shrublands and forests' represents unsuitable areas for skylark breeding (Toepfer & Stubbe 2001), but as they are susceptible to burning, they have potential to become suitable for the species (Table 1). To model the habitat dynamics, transition flows were mediated by temporal rates implicated on the ecological succession of the main structural vegetation covers, including post-fire successions (Moreira *et al.* 2003; section 2.2. of Appendix S2). The fatalities at wind turbines (Fig. 3c) were randomly generated considering the realistic limits based on skylark mortality estimates (Korner-Nievergelt *et al.* 2011) from carcass searches carried out at 10 wind farms in northern Portugal (LEA 2012; section 2 of Appendix S3). The occurrence of biased collision mortality, affecting almost exclusively the adult males (90.9% of the total carcasses found at the study wind farms; Morinha *et al.* 2014), was considered a critical model parameter due to its potential influence on the local population dynamics (section 2.3.3. of Appendix S1). The Local Impact Index (Fig. 3d) was based on the percentage of breeding individuals killed by collisions in each study unit (section 3 of Appendix S4). We used the software STELLA 9.0.3 for the construction of the stochastic dynamic model. The original conceptual submodels (Figs S1–S5), all details and full explanations (Appendices S1–S5), including parameters (Table S1) and equations (Table S2) used in the model construction, are available in Supporting information.

Stochastic dynamic simulations and spatial projections

To estimate the Local Impact Index, every study unit was characterized considering the initial area of each habitat class and the total number of wind turbines, existent and/or expected to be installed during the simulation period. Since the dynamic model simulations are influenced by stochastic factors associated with biological parameters, fire events and mortality attributable to collisions with wind turbines (Appendix S1), 100 independent simulations per study unit were carried out. The timing of the wind turbines' installation was activated throughout the simulations taking as reference 2006, the starting year of the simulations according to the available initial land covers from the Corine Land Cover 2006. Therefore, the spatial representation of the dynamic outputs, projected for each study unit, enabled us to

integrate the geographic projections of the Local Impact Index at a regional level throughout the 20 years simulated. In order to evaluate the regional trend of the Local Impact Index, two time frames were selected, the first year ($t = 1$; year 2007) and the 15th year (year 2021) of the simulation, from which the respective spatial projections were produced. The time frames selected were chosen in accordance with the dynamic modelling requirement of previous available background data, which was based on climatic variables and land cover inventories for 2006 and forecasting scenarios for 2020, also used for the respective projections of skylark breeding distribution (see section "Downscaling of species data and fitting of static species distribution models").

Downscaling and species distribution models

Environmental variables

We first selected a set of variables that, according to expert knowledge, previous reporting in the scientific literature and available data could act as determinants of skylark distribution (Suárez, Garza & Morales 2003; Catry *et al.* 2010). To avoid high correlation between selected variables, we tested for pairwise correlations using Spearman's ρ correlation coefficient, and only variables with correlation lower than 0.7 were considered (Elith *et al.* 2006). In the case of correlated pairs of variables, we chose the one with more direct ecological relevance for the distribution of skylarks (Guisan & Thuiller 2005). This analysis yielded a final set of 14 environmental variables to fit the SDMs (see Table 2 for more details), including attributes of climatic conditions, land cover/landscape composition, fire regime, landscape structure, dispersal corridors and primary productivity.

Downscaling of species data and fitting of static species distribution models

In order to standardize the spatial resolution of the skylark breeding distribution with the pertinent scale to model the species' local population dynamics, we applied a standard technique approach to downscale coarse-scale species data (100 km²), extracted from the Atlas of Breeding Birds of Portugal (Equipa Atlas 2008), into fine-scale species data (1 km²). In this way, the integration of compatible fine-scale spatial projections became adjusted for the quantitative assessment of the impacts caused by wind farms at a regional level (McPherson, Jetz & Rogers 2006). The downscaling procedure used in this work was the centroids approach (Bombi & D'Amen 2012), where the centroids of all occupied 100 km² by the species were considered as presence points. Conversely, we used the centroids of all unoccupied 100-km² cells as absences. The centroids were then used to sample the environmental conditions at the finer resolution (1 km²). The environmental data together with the finer resolution species data (centroids) were used to calibrate and project static species distribution models. Fine-grained predictions were obtained for current conditions and under a climate change scenario for year 2020 (scenarios A1 from the Intergovernmental Panel on Climate Change – IPCC; Nakicenovic & Swart 2000). Static SDMs were calibrated under an ensemble forecasting framework using the BIOMOD2 package (Thuiller *et al.* 2009), available at <http://cran.r-project.org/web/packages/biomod2/index.html> in the free statistical software R (R Development Core Team 2012). Model calibration was performed using the

Table 2. The environmental classes of the variables used in the downscaling procedures for the skylark distribution and the respective main sources of data

Environmental classes	Variables	Source
Climate	AnnPrec (Annual Precipitation)	Current data: http://www.worldclim.org
Landscape composition	pUrb (% cover of urban areas) pAgr (% cover of agricultural areas) pAnCro (% cover of annual crops) pBlFor (% cover of broadleaf forests) pCoFor (% cover of coniferous forests) pMixFor (% cover of mixed forests) pPioMos (% cover of pioneer mosaics)	http://www.eea.europa.eu/data-and-maps/
Fire regime	MaxBurn (maximum percentage of burned area between 1990 and 2009)	http://www.icnf.pt/portal/florestas/dfci/cartografia/info-geo
Dispersal corridors	DensRiv(density of local hydrographic network)	http://sniamb.apambiente.pt
Landscape structure	Alt (altitude) SwiAlt (local diversity of altitude types) NumP (number of patches present in each grid cell)	http://www.cgiar-csi.org/data Patch Analyst
Primary productivity	meanGPP (mean gross primary productivity)	http://www.ntsug.umd.edu/project/mod17

10 available modelling algorithms in BIOMOD2 (for more details, see BIOMOD2 help files and vignettes). The area under the curve (AUC) was obtained as an evaluation output for each model, and only models with AUC > 0.7 were used in the ensemble procedure (Fielding & Bell 1997). The BIOMOD2 package allows the use of a cross-validation repeated split-sample procedure, keeping 20% of the initial data out of the calibration for subsequent validation of the predictions. The number of repetitions was set to 10 (Thuiller *et al.* 2009). Model predictions were then used to produce a single ensemble model applying the *Mean (all)* consensus method (i.e. it decreases the predictive uncertainty of single models by calculating the mean value of the ensemble of predictions), which provides more robust predictions than single models or other consensus methods (Marmion *et al.* 2009). Finally, model projections under current conditions and the climatic scenario considered for year 2020 were reclassified into presence-absence using the ROC threshold (maximizing the percentage of presences and absences correctly predicted, that is the probability where sensitivity = specificity; Liu *et al.* 2005).

CUMULATIVE IMPACT ASSESSMENT AT THE REGIONAL LEVEL

Since the cumulative implications of wind farms on skylark populations are only entirely perceptible when considering regional scales, an integrative assessment of all local impacts on the skylark breeding population dynamics was considered in northern Portugal. Therefore, the likely regional impact on skylarks, induced by wind farms and habitat suitability, was attained by the integration of the outputs from the complementary modelling techniques considered in the proposed framework (Fig. 2 and section ‘The Modelling Framework’ for details). First, the average Local Impact Index for skylarks was calculated based on the overall Local Impact Index values of the study units considered

(500 UTM 1-km² cells) per year of simulation. This metric represents the trend of the average local impact per study unit, considering the incremental load of all wind turbines planned to be installed in the northern region of Portugal through the simulation period. Secondly, the spatially explicit integration of the skylark breeding distribution and the forecasted wind farm impacts (expressed by the Local Impact Index of the study units) was produced for 2007 and 2021, highlighting regional areas with the greatest potential conflicts. Finally, reference population metrics, such as ‘density of breeding pairs per km²’ and ‘density of adults per km²’ (i.e. breeding individuals and floaters excluding juveniles), were calculated from the simulated skylarks’ local population dynamics in contexts without the direct influence of the wind turbines (Appendix S6). These metrics were extrapolated at the regional level as a skylark population abundance background considering only the UTM cells included in the species breeding distribution areas projected for 2007 and 2021. The regional cumulative impact of wind farms on skylarks was therefore inferred by quantifying the total number and percentage of individuals expected to be killed per year by collisions with wind turbines, taking as reference the respective regional estimates for the skylark’s total abundance.

Results

SKYLARK POPULATION DYNAMICS

For demonstration purposes, some possible simulations of skylark population responses to habitat dynamics for one of the study units are shown in Fig. 4. The local abundance of individuals comprises breeders, non-breeders and juveniles, whereas suitable breeding areas include the contributions of natural grasslands, sparsely vegetated areas and burnt areas (Fig. 4a). Skylark population

dynamics are represented by typical fluctuations marked by periods of higher abundance of individuals, during the breeding seasons, and periods of lower abundances in the non-breeding seasons. Additionally, the local population trends are also strongly influenced by the availability of suitable breeding areas mainly determined by a random pattern of fire events (Fig. 4a). Considering these stochastic influences, the average local abundance from 100 independent simulations was projected in order to capture the population trend for each study unit (Fig. 4b).

AVERAGE LOCAL IMPACT INDEX

The obtained results show an increasing average local impact per study unit through the simulation period of 20 years (Fig. 5), where skylark fatalities attributable to the collision with wind turbines are predicted to increase from 1.3% of the local breeding populations in 2006 to 4% in 2026. This trend reflects the intensification effect

of wind farms' gradual installation within the study area, ranging between 412 wind turbines in 2006 (35% of the study units projected to have wind turbines) and 1192 wind turbines in 2014 (100% of the study units with wind turbines), which remain stable until 2026 (Fig. 5).

SPATIOTEMPORAL PROJECTIONS AND MORTALITY ESTIMATIONS AT THE REGIONAL LEVEL

The visual inspection of the spatially explicit dynamic outputs attained for the impact of wind farms on skylark populations, in 2007 and 2021 (Fig. 6), shows that the greatest potential for conservationist conflict is coincident with the ridges of mountainous areas, the species' main breeding habitats. In comparative terms, around 5% of the estimated skylark breeding distribution area was under the influence of wind farms in 2007, increasing to more than 12% in 2021. For the same period, the species breeding distribution area is predicted to decrease 4.5%,

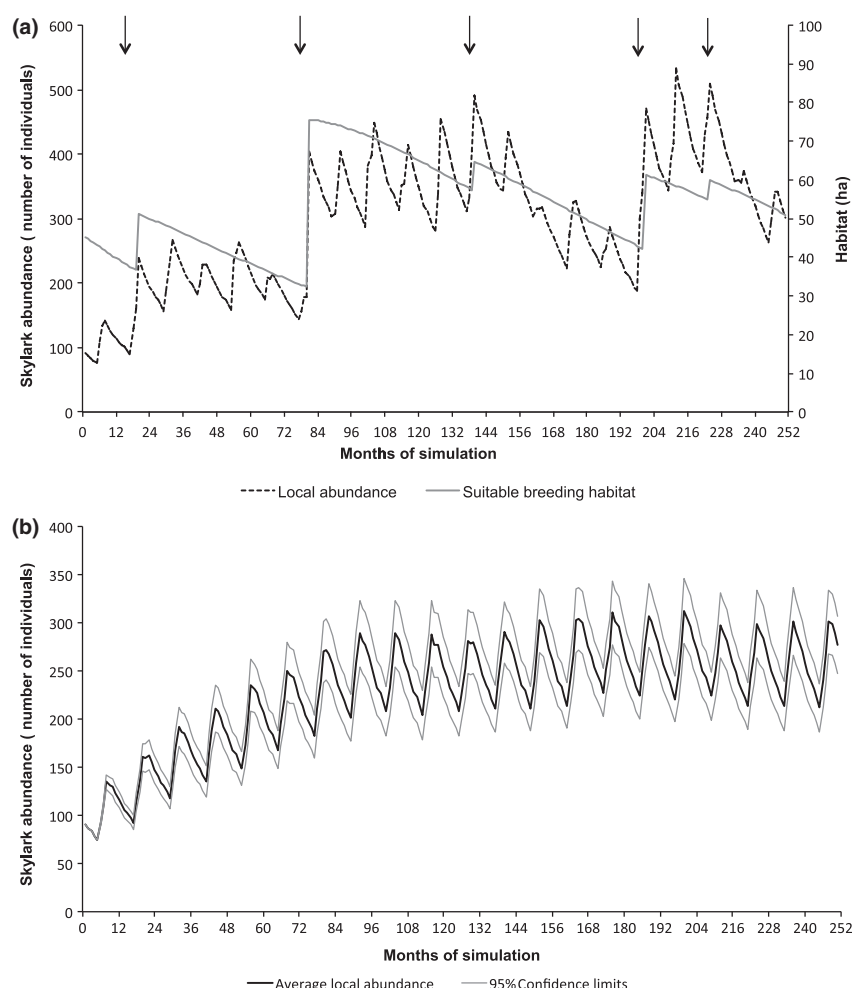


Fig. 4. Possible simulations of the dynamic model concerning skylark population responses to habitat dynamics in one of the study units, throughout a period of 20 years, expressed as: (a) local abundance from a single simulation where the arrows mark the random occurrence of fire events that provide the appearance of new suitable breeding areas and (b) average local abundance with the respective 95% confidence limits from 100 independent simulations.

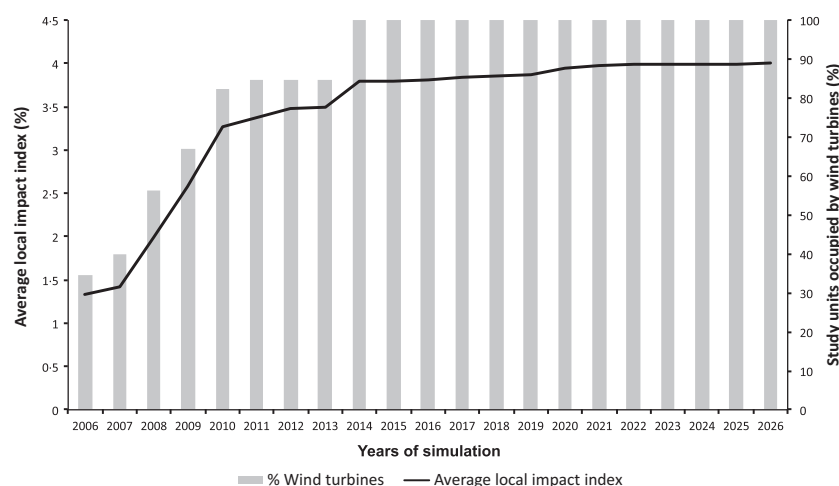


Fig. 5. The average Local Impact Index induced by wind farms on the skylark breeding populations, expressed in mean number of collision fatalities per UTM study unit (1 km²), considering the incremental load of all wind turbines planned to be installed in the northern region of Portugal throughout the simulation period of 20 years.

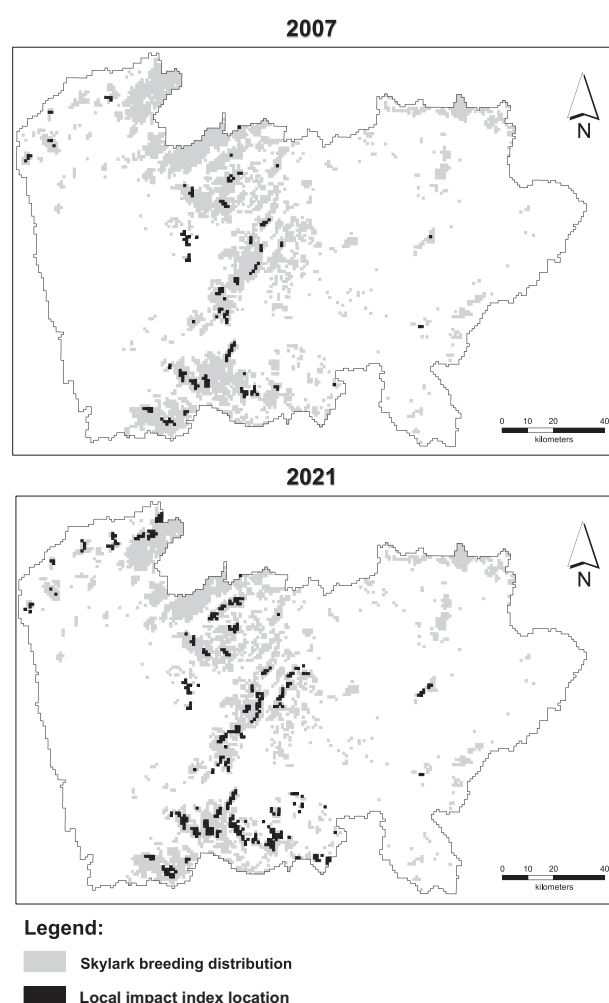


Fig. 6. Spatially explicit integration of skylark breeding distribution, which coincides with the main mountain areas of the study region, and the Local Impact Index location for 2007 and 2021.

from 3314 km² in 2007 to 3165 km² in 2021, as a consequence of the scenario considered for climate and land cover changes. The density of breeding individuals, obtained from the skylark's local population dynamic simulations, indicates an average of 26.8 breeding pairs per km² from a total of 90.6 adults per km². Taking into account these reference values and the species breeding distribution areas projected for 2007 and 2021, the estimated 88 776 skylark total breeding pairs (from a total of 300 267 adults) in 2007 are expected to decrease to 84 785 breeding pairs (from a total of 286 767 adults) in 2021, considering only the effect of the breeding distribution area reduction in northern Portugal. Additionally, the simulated skylark mortality associated with all wind turbines installed at the regional level rises from 1102 fatalities among the breeders in 2007 to 3128 fatalities in 2021, which represents a drastic increase of *c.* 184%. Overall, these combined results represent an intensification of the direct cumulative regional impact for skylark populations, expressed by the percentage of the total number of breeders killed by collisions with wind turbines, from 1.2% in 2007 to 3.7% in 2021 of the total breeding population from northern Portugal.

Discussion

ASSESSMENT OF THE CUMULATIVE IMPACT OF WIND FARMS ON SKYLARKS

Portugal is one of the European Mediterranean countries where the climate is expected to change substantially (EEA 2010), and the consequences on species distributions are assumed to represent a relevant background source of environmental stress, especially when combined with persistent infrastructural impacts (Harrison, Berry & Dawson 2001). In this context, despite the recognition

that the wind farms and the associated local ecological impacts represent a current cause for concern (Masden *et al.* 2010), there is still a lack of evaluation and forecasting of their long-term cumulative ecological consequences (Roscioni *et al.* 2013, 2014; Santos *et al.* 2013). These are, for instance, the impacts resulting from the intensification of wind farm installation on the skylark breeding populations, one of the most affected bird species in Portugal by direct collision with wind turbines (Bernardino *et al.* 2010; LEA 2012). Moreover, the skylark breeding distribution area was predicted to decrease taking into consideration the future environmental changes projected for 2021, which might rely on the expected shift towards warmer conditions in mountain regions (Theurillat & Guisan 2001). In northern Portugal, skylarks preferentially select habitats at higher altitudes to breed and the predicted population decline may reflect important environmental constraints for breeders, namely in terms of habitat loss and/or deterioration of habitat quality (Sanz-Elorza *et al.* 2003; Drewitt & Langston 2006). The expected installation of new wind farms aggravates this scenario, considering that more than 12% of the estimated skylark breeding distribution area in 2021 will be under the influence of these infrastructures. Since skylark collision mortality was predicted to have a radical increase for the same period at the regional level (about 184%), this seems to represent a non-negligible persistent impact with potential consequences on the size and quality of the affected populations (Drewitt & Langston 2006). In fact, the combination of these detrimental trends with the sex-biased mortality reported for the study area (Morinha *et al.* 2014) might have consequences beyond the known direct impacts, namely in the reproductive success and fitness of the local populations (Bjørnstad, Ims & Lambin 1999).

Our reference outputs are in agreement with field observations and other studies that investigated the ecological consequences of similar anthropogenic and environmental changes (Harrison, Berry & Dawson 2001; Pearce-Higgins *et al.* 2008). For example, the skylark average density of 26.8 breeding pairs per km² (from a total of 90.6 adults per km²), simulated for natural grasslands and sparsely vegetated habitats from the uplands of northern Portugal, is within the interval of values reported by several European studies for the species in natural grasslands, ranging from 11 to 90 breeding pairs per km² (Donald 2004). Moreover, in a mountainous area from Galicia (north-west Spain), a contiguous region of the study area with similar biogeographic characteristics and global extent, the average skylark total abundance was estimated at 269 955 adults (breeding and non-breeding birds, excluding the juveniles) for 2004–2006, ranging between 195 775 and 354 127 birds (Carrascal & Palomino 2008), compatible with our simulated value of 300 267 adults for the study area in 2007, which seems to represent a credible modelling performance.

IMPLICATIONS FOR PROJECT-BASED AND STRATEGIC MONITORING

Overall, the spatially explicit regional patterns of the metrics simulated seem to capture with credibility the effects of the intensification of wind farm installation combined with scenarios of environmental changes towards new expected conditions. These outputs are encouraging since they seem to represent a step forward in evaluating the regional consequences of wind farms on a vulnerable bird species, but are easily adaptable to other species and circumstances. This kind of new model-based methodology could be used to support strategic options for impact mitigation and management, for example in the context of project-based impact assessments, by providing projections of long-term indicator trends under realistic social–ecological change scenarios (Topping, Odderskær & Kahlert 2013). Additionally, this could act as an integrative multi-scale modelling tool to guide and improve the strategic monitoring of wildlife change driven by combined sources of pressures, providing insights into future data gathering efforts for the evaluation of cumulative impacts on biodiversity (Roscioni *et al.* 2013; Santos *et al.* 2013). The proposed framework could thus be integrated in the paradigm of wider biodiversity observation networks (e.g. LTER, GEO-BON) to reinforce the quality of long-term impact assessments (Scholes *et al.* 2008) by improving the potential applicability of the existing data bases. Moreover, the capacity to accurately forecast responses of standard ecological indicators to landscape changes is crucial considering the distinct drivers that act on vulnerable species, guilds and communities. Since modelling expertise is required, our framework is particularly suitable for management recommendations in the scope of conservation programmes, namely by anticipating, with scientific credibility, future ecological consequences associated with infrastructural impacts. In this perspective, we highlight the interplay between model-based research and biodiversity monitoring, which will make the methodology more instructive and credible to technicians, decision-makers and environmental managers. In fact, predictive modelling tools can contribute to an increasing efficiency and usefulness of monitoring results for assessing and mitigating anthropogenic impacts, whereas strategic monitoring can provide robust data sets to validate models and improve their predictive power (Brotons, Herrando & Pla 2007).

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Data accessibility

Data are available in the online supporting information.

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Supporting Information

Additional Supporting Information may be found in the online version of this article.

Figure S1. Conceptual diagram of the sub-model for simulating the skylark population dynamics.

Figure S2. Conceptual diagram of the sub-model for simulating the habitat dynamics.

Figure S3. Conceptual diagram of the sub-model for simulating skylark fatalities at wind turbines.

Figure S4. Conceptual diagram of the sub-model for calculating the Local Impact Index.

Figure S5. Conceptual diagram of the sub-model for calculating independent metrics of skylark breeding productivity.

Table S1. Specification of the main variables included in the model and respective description, units and bibliographic source.

Table S2. Specification of all mathematic equations included in the model.

Appendix S1. Detailed conceptualization description of the sub-model to simulate skylark population dynamics.

Appendix S2. Detailed conceptualization description of the sub-model to simulate the habitat dynamics.

Appendix S3. Detailed conceptualization description of the sub-model to simulate skylark fatalities at wind turbines.

Appendix S4. Detailed conceptualization description of the sub-model to calculate the Local Impact Index.

Appendix S5. Detailed conceptualization description of the sub-model to calculate independent metrics of the skylark breeding productivity.

Appendix S6. Detailed description of the metrics estimates “density of breeding pairs per km²” and “density of adults per km²”.