



Prioritizing road-kill mitigation areas: A spatially explicit national-scale model for an elusive carnivore

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Abstract

Aim: Roads impact wildlife in different ways, among which road mortality has been the most studied. Budgets in conservation biology are usually small, and macroecological approaches have been employed in recent years as the first steps towards guiding management. Carnivores are particularly vulnerable to mortality on roads due to their elevated ecological needs (low population density, often low fecundity and relatively large home ranges). Our aim was to develop a ranking methodology to prioritize specific areas for road-kill mitigation.

Location: Continental Italy.

Methods: We studied 271 occurrences of live polecats (*Mustela putorius*) and 212 polecat road-kill sites. We used the former to generate a species distribution model and the latter to identify the variables that determined the road-kill risk. Habitat suitability was derived from a spatial distribution model that combined the polecat occurrence data with a set of environmental variables. Prey availability was derived from the combination of suitability maps of 26 prey species. We used generalized linear modelling to identify the set of variables that best explained the occurrence of road-kills. The variables included in the best performing model were combined to produce the road risk map and to identify the areas with the highest densities of road sections with highest risk.

Results: Road-kills were positively associated with the road sections with higher broad-leaved forest coverage. The number of casualties was found to be higher than expected on the national and provincial roads and lower than expected on the local roads.

Main conclusions: This approach allowed us to identify the 10 × 10 km cells where mitigation actions to prevent road-kills should be prioritized. As mitigation actions (wildlife passage construction, fencing) are expensive, measures should be prioritized on the specific high-risk road sections inside these selected cells, avoiding generalized mitigation plans.

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KEYWORDS

citizen science, Italy, *Mustela putorius*, prioritization, Road Ecology, road-kill, species distribution model

1 | INTRODUCTION

The increase in global road network has an enormous impact on both the environment and wildlife (Forman et al., 2003; Van der Ree, Gagnon, & Smith, 2015). Road Ecology shows that this linear infrastructure can impact wildlife through several mechanisms (Coffin, 2007; Forman & Alexander, 1988). For instance, roads can act as ecological barriers that reduce the movement of the species and alter, degrade and fragment the surrounding habitats (Bhattacharya, Primack, & Gerwein, 2003; Coffin, 2007; Hawbaker & Radeloff, 2004). This habitat fragmentation can cause segregation of populations, leading, in extreme cases, to their genetic differentiation (Balkenhol & Waits, 2009; Riley et al., 2006) or even extinction (Ceia-Hasse, Borda-de-Água, Grilo, & Pereira, 2017). However, the most visible and studied impact of roads are wildlife–vehicle collisions (Coffin, 2007; D'Amico, Ascensão, Fabrizio, Barrientos, & Gortázar, 2018; Rytwinski & Fahrig, 2015). Carnivores are especially vulnerable to road fatalities due to their large home ranges, low population densities and often low reproductive rates (Ceia-Hasse et al., 2017; Ginsberg, 2001; Grilo, Bissonette, & Santos-Reis, 2009). Moreover, as small mammals often take shelter on road verges (reviewed in Ascensão, Lapoint, & Van Der Ree, 2015), predators hunting in the proximity of roads can be killed on roads at high rates (Barrientos & Bolonio, 2009; Planillo, Mata, Manica, & Malo, 2018). Certain predators also use roads or railway tracks during displacements (Latham, Latham, Boyce, & Boutin, 2011), a behaviour that increases their road-kill rates (Kaczensky et al., 2003).

Budgets for conservation are usually constrained. In recent years, macroecological approaches that have been frequently based on open-source data, such as citizen science projects, have been increasingly employed to guide managers before implementing expensive mitigation actions. For instance, Brehme, Hathaway, and Fisher (2018) used macroecological approaches to rank the reptiles and amphibians most susceptible to road mortality and fragmentation in California based on their life histories and space-use characteristics after a literature review. In a spatially explicit approach, González-Suárez, Zanchetta Ferreira, and Grilo (2018) explored how life history traits could explain road-kill risk for birds and mammals in Brazil. On the other hand, Beston, Diffendorfer, Loss, and Johnson (2016) developed a prioritization system to identify the avian species most likely to experience population declines from wind facilities based on their current conservation statuses or population sizes and their expected risk from fatalities due to collision with turbines. Other authors like D'Amico et al. (2019) used atlas data and species traits from the literature to rank species based on their risk of population extinction from collision with power lines and to identify the areas where high richness of these species overlapped with current electricity grids. In all cases, these approaches were the first step

before implementing field-level mitigation actions, such as fencing, underpass construction or wire marking.

Here, we present a method aimed at identifying areas with greater concentrations of road sections with high risk for road-kills at a large scale, with the elusive polecat (*Mustela putorius*) as the case study and continental Italy as the study area. For this declining species (Croose et al., 2018), road mortality represents one of its most important threats (Barrientos & Bolonio, 2009; Barrientos & Miranda, 2012; Croose et al., 2018). Specifically, we aimed to (a) identify the variables, both habitat- and road-related, determining the polecat road mortality in Italy; (b) generate a spatially explicit road-kill risk map for the polecat at the national level; and (c) identify the top-ranking cells concentrating road sections where mitigation to avoid road-kills should be prioritized.

2 | METHODS

2.1 | Study species

The polecat is a small (500–1,100 g) carnivore belonging to the Mustelid family (Blandford, 1987; Croose et al., 2018). This species has a Palearctic distribution, extending from Western Europe to the Urals (Blandford, 1987; Croose et al., 2018). In recent years, the polecat has undergone a decline in most of its distribution range, which has been attributed to excessive hunting pressure, poisoning and the loss of preferred habitats (Croose et al., 2018). Moreover, the decline in prey populations, such as rabbits and amphibians, has influenced the population crashes in some regions (Baghli & Verhagen, 2003; Birks & Kitchener, 1999; Roger, Delattre, & Herrenschmidt, 1988). Competition with the invasive American mink (*Neovison vison*) (Barrientos, 2015) and hybridization with feral ferrets (*M. p. furo*) (Costa et al., 2013; Croose et al., 2018) have been recently identified as additional threats. As mentioned above, one of the main causes of its current decline is road mortality (Blandford, 1987; Croose et al., 2018; Virgós, Cabezas-Díaz, & Lozano, 2007), which is likely higher because this species uses road verges as hunting areas (Barrientos & Bolonio, 2009; Barrientos & Miranda, 2012; Blandford, 1987).

2.2 | Summary of the modelling framework

The map of road risk for polecats was built by combining several raster maps, each describing a specific factor that could affect polecat collision risk for this species. These included road characteristics and ecologically related factors like habitat suitability and prey availability and five environmental factors related to the matrix permeability to polecat movements (Figure 1). Habitat

FIGURE 1 This flow chart summarizes the various procedures used to create the road risk model for the polecat in Italy. CLC, Corine Land Cover; GLM, generalized linear model; NEN, National Ecological Network; SDM, species distribution model; DEM, digital elevation model

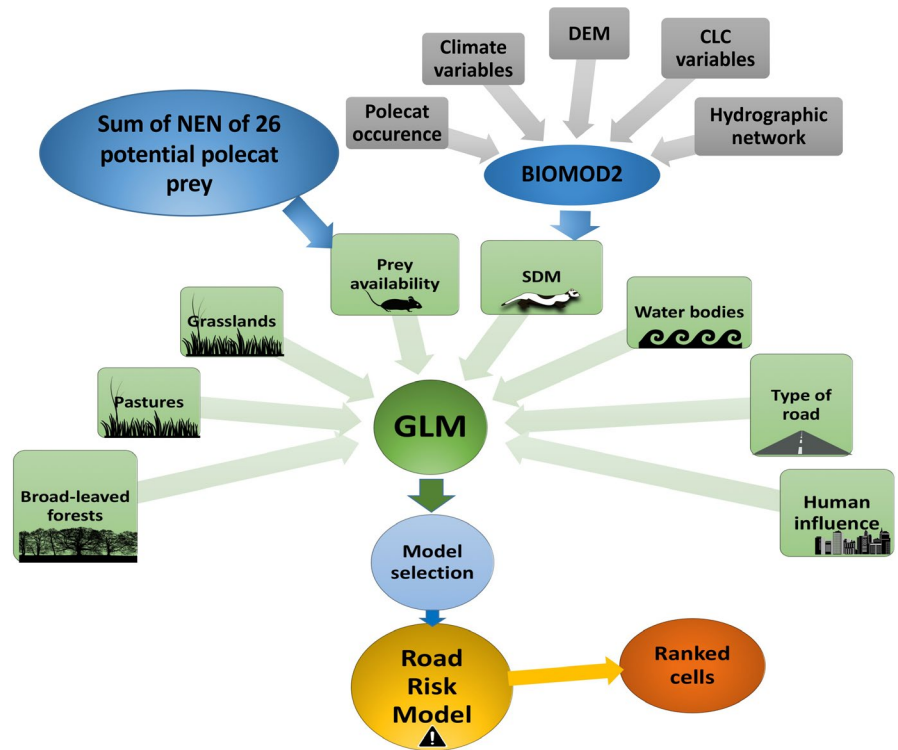
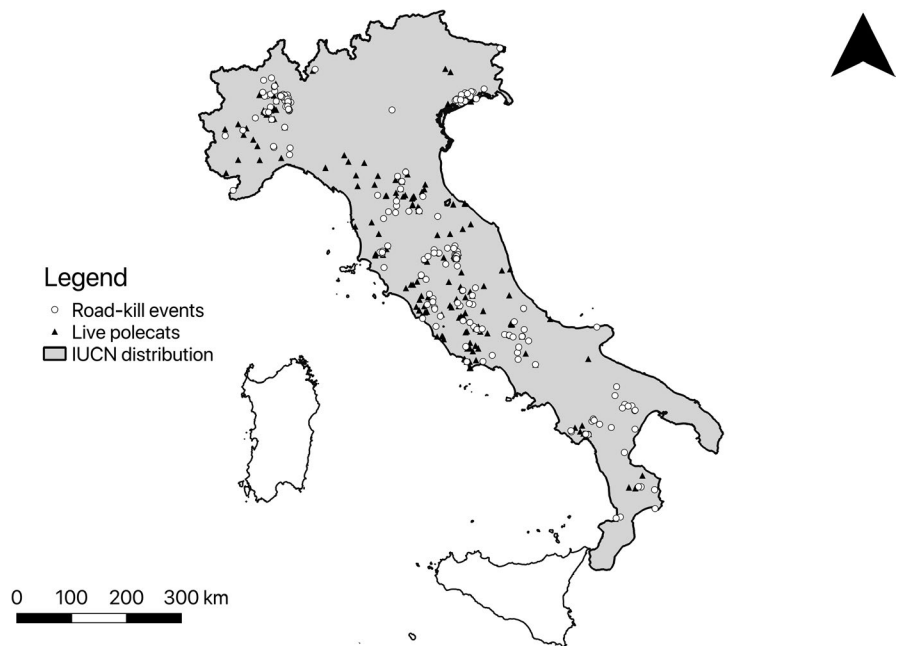


FIGURE 2 Locations of the polecat occurrences (black triangles; $n = 271$) used to generate the SDM, and the polecat road-kill events (white circles; $n = 212$) used to produce the road risk model. Data are mapped on the Italian range of the species based on IUCN (Skumatov et al., 2016)



suitability was derived from a spatial distribution model that combined the polecat occurrence data with a set of environmental variables related to polecat biology. Prey availability was derived from the combination of available expert-based suitability maps of 26 preferred prey species. The relationships among the road risk locations and the eight variables were analysed with generalized linear models (GLM). The variables included in the best performing model were then combined to produce the road risk map and to identify the areas with the highest densities of risky road sections (see Figure 1 for flow chart).

2.3 | Road-kill and presence data

The locations of road-kill incidents ($n = 212$) and live individuals ($n = 271$) (Figure 2) were obtained from regional authorities, literature (Bizzarri, Lacrimini, & Ragni, 2010; Bon, 2017; Fusillo & Marcelli, 2014; Ragni et al., 2014; Rondinini, Ercoli, & Boitani, 2006), researchers, NGOs and from those citizen science projects (Appendix S1) providing data validated by experts. Moreover, we used only data that referred to direct observations and discarded records of indirect signs, such as scat or footprints. Previous studies have shown

the contribution of citizen science to a number of research topics, including the study of road fatalities (Barrientos & Miranda, 2012; Heigl, Horvath, Laaha, & Zaller, 2017; Périquet, Roxburgh, le Roux, & Collinson, 2018; Tiago, Pereira, & Capinha, 2017). The road-kill dataset included 60 records in which the specimen was found on the road, although it was not explicitly reported as a road casualty. The dataset covered the time spans of 1990 to 2017 for road-kills and 1968 to 2017 for direct observations, considering that the road network has undergone few changes in these years (Pinto & Franchin, 2010). However, the oldest data represented only a small proportion of the dataset (road-kills: 1990–1999 = 7%; direct observations: 1968–1999 = 8%).

2.4 | Habitat suitability

We used the 271 live polecat records to build a species distribution model (“SDM”, hereafter) for the polecat in Italy. As there is a strong correlation between the road-kill numbers and the species abundance (D’Amico, Román, de los Reyes, & Revilla, 2015; Møller, Erritzøe, & Erritzøe, 2011; Santos et al., 2016), we used the relative probability of occurrence in each grid cell as a surrogate for the species abundance in the surroundings of the road-kill site (see Visintin, van der Ree, & McCarthy, 2016 for a similar approach). Frequently, occurrence datasets derived from heterogeneous sources (e.g. citizen science initiatives and museums) suffer from sampling biases, but it is difficult to assess the effect of these biases on model predictions. Such biases can lead to environmental bias as well by overrepresenting the environmental conditions associated with regions of denser sampling (e.g. Anderson & Gonzalez 2011). Accordingly, we used “spThin,” which is an R package that has been employed in several SDM studies (e.g. Febbraro et al., 2019; Ramesh, Gopalakrishna, Barve, & Melnick, 2017), to reduce spatial autocorrelation in the occurrence records (Aiello-Lammens, Boria, Radosavljevic, Vilela, & Anderson, 2015). After this filtering procedure, we reduced the final dataset for constructing the SDM to 172 polecat occurrences. To build the models, we considered the Corine Land Cover at III level (CLC, 1:100,000; 2012), along with the presence/absence of water bodies (derived from hydrographic network 1:250,000), a digital elevation model and a set of climate data. The last two datasets were obtained from the Worldclim database (with a 30 arc-seconds resolution, Fick & Hijmans, 2017). Climate and elevation predictors were checked for multicollinearity by calculating Pearson’s coefficient. Only the variables with $r < .7$ were retained. The final set of predictors that entered the model were as follows: annual mean temperature (BIO1), mean diurnal range (BIO2), temperature seasonality (BIO4), temperature annual range (BIO7), mean temperature of wettest quarter (BIO8), annual precipitation (BIO12), precipitation of warmest quarter (BIO18), rainfall of coldest quarter (BIO19), elevation, CLC and hydrographic network. Models were performed through an ensemble forecasting approach as implemented in Biomod2 platform within R (Thuiller, Lafourcade, Engler, & Araújo, 2009). Specifically, we included the following seven algorithms into

the ensemble modelling approach: general linear models, generalized additive models, generalized boosted models, classification t analysis, artificial neural network, multiple adaptive regression splines and random forest models (see Appendix S1 for the model calibration). We randomly splitted each the occurrence dataset into an 80% sample for model calibration and 20% for model validation; the procedure was repeated ten times, and the results were averaged (Russo et al., 2014). The predictive performance of the models was assessed by measuring the area under the receiver operating characteristic curve (AUC; Hanley & McNeil, 1982) and the true skill statistic (TSS; Allouche, Tsoar, & Kadmon, 2006). To avoid using poorly calibrated models, only the models with $AUC \geq 0.70$ were considered in the subsequent analyses (Di Febbraro, Martinoli, Russo, Preatoni, & Bertolino, 2016). The models were averaged by weighting the individual model predictions by their AUC scores and averaging the result (Marmion, Parviainen, Luoto, Heikkinen, & Thuiller, 2009).

2.5 | Prey availability

Because “prey availability” may represent a variable associated with road-kills (Barrientos & Bolonio, 2009; Barrientos & Miranda, 2012), we considered the habitat suitability models provided by the National Ecological Network (NEN) project (Boitani, Falcucci, Maiorano, & Montemaggiore, 2003) for 26 potential polecat prey species (Baghli, Walzberg, & Verhagen, 2005; Lodé, 1997; Prigioni & De Marinis, 1995; Sidorovich & Pikulik, 1997; see Appendix S1). The NEN rasters are provided at a resolution of 100×100 m grid cells, where each cell has a suitability score from 0 (unsuitable) to 3 (highly suitable). The 26 suitability maps were combined into a single raster by summing the suitability score of each cell. This map was then used as a proxy for prey availability in a 50 m buffer radius generated around the random and road-kill points.

2.6 | Road risk model

To produce the road risk model (RRM), we related a set of environmental characteristics with 212 road-kill locations and an equal number of randomly chosen points along the road network. One random point per road-kill was generated within a section of road included within a circular buffer, with a radius of 1.5 km, centred on the corresponding road-kill. We selected this radius because it is the average home range size for polecats (Baghli et al., 2005; Rondinini et al., 2006) and therefore represents a conservative approach for selecting control points within the potential polecat habitat. Within each buffer, we recorded the SDM value and the prey availability value. Also, we used “type of road” as a categorical variable because it is related to road-kill rates (Barrientos & Miranda, 2012). It had three levels, that is, “state,” “provincial” and “local” roads. State roads are the main traffic routes, have speed limits of 90–100 km/hr and are often bounded by barriers (e.g. Jersey barriers). Provincial roads connect various cities in the

same region and have speed limits of approximately 70 km/hr. Finally, local roads are urban roads with speed limits of 50 km/hr. All types of roads are usually two-lane roads. Motorways were not considered due to their lack of representation in the data (only one random point and three road-killed polecats were found along motorways). Since urban habitats are positively related to carnivore road casualties (Barrientos & Bolonio, 2009; Grilo et al., 2009) and can be positively selected by polecats (Baghli et al., 2005; Virgós, 2003; Zabala, Zuberogoitia, & Martínez-Climent, 2005), we calculated "human influence" as the distance in metres to the nearest town and divided it by the population density (ISTAT, 2011); accordingly, surroundings of more populated towns have greater weight than less populated ones. As polecats are known to move across specific habitat types (Baghli et al., 2005), we used the most relevant land cover category within a buffer of 50 m radius around each point to determine the coverage of these preferred habitats, that is, "grasslands" (CLC 231), "pastures" (CLC 321) and "broad-leaved forests" (CLC 311), as a proxy of the landscape structure (Barrientos & Bolonio, 2009). In case the buffer did not fall into one of these categories, the value zero was assigned. We used the 1990 CLC (scale 1:100,000) inventory for road-kills from 1990 to 1999 and the 2000 CLC inventory for road-kills from 2000 to 2017. Finally, as polecats are also known to move preferentially along shores (Mestre, Ferreira, & Mira, 2007; Rondinini et al., 2006; Zabala et al., 2005), we measured the distance in metres to the nearest lake or river (hereafter, "water bodies"). All of these spatial analyses were performed using the program QGIS (QGIS Development Team, 2017).

2.7 | Statistical analyses

We conducted a regression analysis on the set of eight environmental variables using a binomial distribution (road-kill points vs. random points) and logit link function. We used the best subset procedure and the Akaike information criterion (AIC) to identify the set of models (i.e. combination of variables) that best explained the occurrence of road-kills. We used this technique because it yields consistent results regardless of the order in which variables are included in the model and allows models with different numbers of parameters to be directly compared with each other (Burnham & Anderson, 2002). We used AIC values without correction for small sample size because the number of observations was high (i.e. the ratio between the number of observations and explanatory variables was over 40) for the number of explanatory variables (Burnham & Anderson, 2002). The models with the lowest AIC values represent the best compromise between the maximal fit and the minimal number of explanatory variables (i.e. statistical parsimony). To evaluate the relative explanatory power of the competing best models, the Akaike weights (ω_i) were calculated. The evidence ratio was calculated to compare the Akaike weights of the best model and the competing ones (Burnham & Anderson, 2002). To estimate the relative importance of every variable included in any of the best models

(those with $\Delta AIC \leq 2$), we calculated the sum of the Akaike weights of the models that included these variables (Burnham & Anderson, 2002). The quantitative differences in the explanatory variables between stretches with and without road-kills were evaluated with paired *t* tests. We analysed the differences in the proportion of road-kills per type of road by using 2×2 contingency tables with the Yates correction. All analyses were performed with the Statistica v.10 software (StatSoft).

We first obtained a subset of the models that best separated the road-kills from the random points (Table 1). We assessed the predictive performance of the two models with best AIC values with the same procedure used for SDM (see above). We then compared the AUC values of these two models using the Wilcoxon-Mann-Whitney and selected the model with highest values of the AUC. We produced a spatially explicit prediction of the road risk at a 100×100 m resolution. Finally, we overlapped the Italian Universal Transverse Mercator 10×10 km grid with the RRM and ranked the cells based on their summed road risk values.

3 | RESULTS

The SDM (Appendix S1) showed fair and poor levels of predictive performance, with AUC values ($M \pm SE$) of 0.732 ± 0.022 and TSS values of 0.371 ± 0.031 .

The range of values of explanatory variables was 0–7,725 m² for grasslands cover, 0–6,707 m² for pastures cover, 0–7,725 m² for broad-leaved forests cover, 0–8,669 m for water bodies, 0–42 for the sum of the NEN suitability scores, 0–2,591 m/habitants for towns and 0–0.734 probability of polecat occurrence based on the SDM.

The regression procedure with the best subset analysis provided a set of twelve similarly plausible models according to their AIC values (i.e. the difference between their AIC and the lowest one was <2; Table 1). More importantly, two variables (broad-leaved forests and type of road; Table 2) were included in all twelve models and, consequently, had the highest weights ($\sum \omega_i = 1.0000$). Specifically, road-kills were positively associated with road sections with higher broad-leaved forest coverage ($M \pm SE$) (road-kill = $1,459 \pm 195$, random = 656 ± 137 ; paired *t* test = 4.13; *df* = 212; *p* < .0001) and were found at higher-than-expected rates on state ($n_{\text{road kills}} = 87$, $n_{\text{random}} = 28$; $\chi^2 = 15.06$; *df* = 1; *p* < .001) and provincial roads ($n_{\text{road kills}} = 73$, $n_{\text{random}} = 31$; $\chi^2 = 8.02$; *df* = 1; *p* < .01) and at lower-than-expected rates on local roads ($n_{\text{road kills}} = 49$, $n_{\text{random}} = 152$; $\chi^2 = 27.10$; *df* = 1; *p* < .0001). The SDM ($\sum \omega_i = 0.5038$) or prey availability ($\sum \omega_i = 0.1859$) were less influential.

The two top-ranked models in Table 1 showed fair to good levels of predictive performance. Specifically, the two top-ranked models had AUC = 0.766 (*SD* = 0.290) and AUC = 0.777 (*SD* = 0.305), and TSS = 0.475 (*SD* = 0.0572) and TSS = 0.483 (*SD* = 0.579), respectively. The two models significantly differed in their predictive performance (Wilcoxon *W* = 3,804, *p* = .003). Thus, we used the second one as the best performing model to produce the spatially explicit risk model.

TABLE 1 List of the twelve models that best separated the road-kills from the random points. The Δ AIC was the difference in the AIC values compared to the estimated best model (lowest AIC) that allowed the ranking of models from an estimated best (top of the table) to worst. Akaike weight is the estimated probability that a model is the best model in the set. The evidence ratio indicates to what extent the best model is better than another. The K is the number of predictors contained in the model. See the full set of models in Appendix S1

Model no.	Variables contained in the model	K	Δ AIC	Akaike weight (ω_i)	Evidence ratio
1	Broad-leaved forests + Type of road	2	0	0.150	0
2	Broad-leaved forests + SDM + Type of road	3	0.041	0.147	2
3	Broad-leaved forests + Prey availability + Type of road	3	0.874	0.097	55
4	Water bodies + Broad-leaved forests + SDM + Type of road	4	0.998	0.091	65
5	Broad-leaved forests + Prey availability + SDM + Type of road	4	1.044	0.089	68
6	Water bodies + Broad-leaved forests + Type of road	3	1.454	0.073	107
7	Broad-leaved forests + Grasslands + SDM + Type of road	4	1.781	0.062	144
8	Broad-leaved forests + Grasslands + Type of road	3	1.798	0.061	146
9	Broad-leaved forests + Pastures + SDM + Type of road	4	1.875	0.059	155
10	Broad-leaved forests + Pastures + Type of road	3	1.887	0.058	157
11	Human influence + Broad-leaved forests + Type of road	3	1.921	0.057	161
12	Human influence + Broad-leaved forests + SDM + Type of road	4	1.954	0.057	166

Abbreviations: AIC, Akaike information criterion; SDM, species distribution model.

TABLE 2 The best model after regression analysis with the best subset procedure following the AIC criteria. The not standardized regression coefficients were used to determine the risk values of each 100×100 m cell

Variable	Category	Regression coefficients
Intercept		-0.676
Broad-leaved forests		0.001
Type of road	State	2.367
Type of road	Provincial	1.935
SDM		-0.002

Abbreviations: AIC, Akaike information criterion; SDM, species distribution model.

Figure 3 shows the 10×10 km cells based on their road risk values. The risk value was computed by summing up RRM values of road sections within the cell. High-risk cells were concentrated in North and West Italy (Figure 3). Finally, Figure 4 shows in detail one of these top-ranked cells, with those sections where mitigation should be prioritized within the cell highlighted in red colours.

4 | DISCUSSION

The approach used in the present study enabled for the first time testing of how environmental variables affected road-kill risk at a

large scale, which led to a ranking of the areas with high numbers of risky road sections based on relatively easily accessible information. This method was empirically supported as it was based on the traits that differentiated the road sections with recorded road-kills from those without. Since the mitigation actions to prevent road-kills, such as wildlife passage construction or fencing, are expensive (Rytwinski et al., 2016; Smith, van der Ree, & Rosell, 2015; Van der Ree et al., 2015), our macroecological approach represents a promising tool for selecting road sections where these mitigation actions should be implemented, thus reducing costs (e.g. generalized mitigation). The 10×10 km cells were ranked by their road-kill risk, which was determined by the total length of risky road stretches within the cells, that is by the abundance of broad-leaved forests bisected by national or provincial roads within suitable polecat distribution range. Once focused on those cells with higher concentrations of potentially high-risk road sections, the mitigation actions should be prioritized in the road sections with the highest potential risk (those marked in red colours in the example from Figure 4). We would like to add a cautionary note to our model as some landscape traits can be captured with a low resolution by a 50 m buffer. Our approach will become more accurate in the future if fine-scale environmental data become available. However, we chose to use this buffer size as it is the most accurate for achieving truly effective mitigation actions at local scales (Barrientos & Bolonio, 2009; Barrientos & Miranda, 2012).

The importance of broad-leaved forests seems to be related to landscape structure and animal movements. This kind of forest

FIGURE 3 Road-kill risk map for the polecat in Italy. Cells (10 × 10 km) were ranked based on their summed road risk

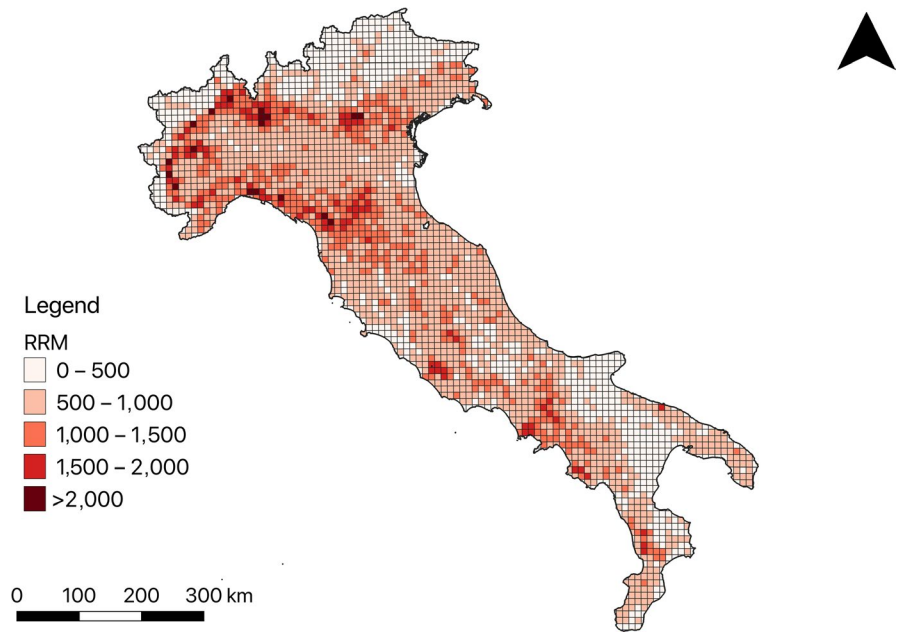
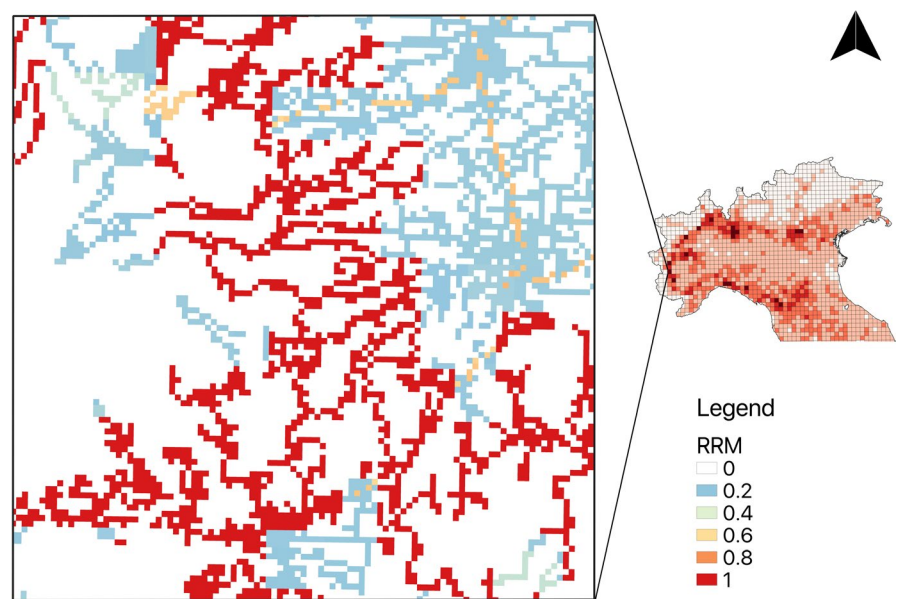


FIGURE 4 Enlargement of one of the cells with the highest concentrations of high-risk road sections (in red) for road mortality of polecat in Italy



includes riparian and gallery woodlands. These ecosystems were selected in other areas by polecats, such as in Luxemburg, Spain, Italy or Portugal, probably because the dense stream margin vegetation provides protection during their movements (Baghli et al., 2005; Mestre et al., 2007; Rondinini et al., 2006; Zabala et al., 2005). This is a general pattern for all carnivores, especially in agriculture-dominated matrixes (Virgós, 2001). Riparian habitats also provide a variety of food resources because they act as ecotones, which increases their value for polecats as hunting areas (Mestre et al., 2007; Zabala et al., 2005). Beech and oak forests were also used at above-average rates in the study of Baghli et al. (2005), especially in summer. Although we do not know the sex of the casualties in our study, road-kill datasets are usually male-skewed (Barrientos, 2015). Interestingly, the preference to move across deciduous forests seems stronger in

males (Baghli et al., 2005), likely because males usually move for longer distances in search of receptive females and are more prone to dispersal (Rondinini et al., 2006). A significant relationship between the type of road (highly related to traffic flow) and road-kill rates has been frequently found, including in studies on carnivores (Clarke, White, & Harris, 1998; Philcox, Grogan, & Macdonald, 1999; Seiler, 2005). Namely, roads with low traffic levels (i.e. local roads) do not represent a threat compared to other types of roads, likely because they usually have little traffic that moves at low speed; on the contrary, roads with both higher traffic and speed limits (i.e. provincial and national roads) present higher road-kill rates. Finally, the high traffic intensity on motorways discourages wildlife from attempting to cross them, although we do not have enough data to validate this last conclusion. However, it is also possible that the population

declined in these areas due to previous road mortality (Ascensão et al., 2019; Teixeira, Kindel, Hartz, Mitchell, & Fahrig, 2017).

The importance of potential prey did not determine the occurrence of road-kills in Italy, as happens, for instance, in Mediterranean Spain (Barrientos & Bolonio, 2009; Barrientos & Miranda, 2012). This could be because a single prey species (the European rabbit *Oryctolagus cuniculus*) is by far the main prey in Spain (Barrientos & Bolonio, 2009), which makes it easier to model the influence of prey abundance on the probability of road-kill. Alternatively, the field sampling carried out by Barrientos and Bolonio (2009) could be more accurate than the map of habitat suitability for prey species (Boitani et al., 2003) that we used in the present study.

Finally, our study emphasized the potential utility of citizen science as a first step in conservation programmes, especially when budgets are limited, since we have used this open-source data for our analyses. Citizen science projects are increasingly becoming a common source of data in road-kill studies (e.g. Fabrizio, Di Febbraro, D'Amico, et al., 2019; Heigl et al., 2017; Santori, Spencer, Van Dyke, & Thompson, 2018), often taking advantage of the development of new technologies such as mobile recording devices (reviewed in Shilling, Perkins, & Collinson, 2015). However, citizen scientists (i.e. occasional, often non-trained reporters) who gather road-kills in a sporadic way can be biased towards larger species compared to trained patrols or scientists (Périquet et al., 2018). Additionally, small carcasses such as those of mustelids persist on the road for a shorter period and are detected at lower rates compared to those of large mammals (Barrientos et al., 2018). Those carcasses that are repeatedly run over are flattened, becoming unattractive for citizen scientists, who lose motivation or put less effort into reporting unidentified carcasses or non-flagship species (Lukyanenko, Parsons, & Wiersma, 2016). These biases might have affected the unequal distribution of records of live polecats in the study area, that is the small number of records collected in southern Italy, while the number of road-kill incidents remained high in this region. This in turn might explain the low predictive performance of the SDM in our study. However, according to Schwartz, Shilling, and Perkins (2020) and to our experience with other mustelids (Fabrizio, Di Febbraro, D'Amico, et al., 2019; Fabrizio, Di Febbraro, & Loy, 2019), this claims for increasing efforts in collecting road-kill data on small-medium mustelids at country scale. We stress the relevance of considering both the RRM and the potential distribution of the target species as the most appropriate method for prioritizing road sections in need of mitigation measures.

Despite the utility of citizen science to identify priority areas for mitigation, we should not lose sight of the fact that the final step of any road-kill study should be to apply the corresponding mitigation measures in the field. The implementation, and success in reducing road-kills, of these measures (fencing, construction of wildlife passages) will be the definitive proof that our method has worked. Finally, medium- to long-term monitoring is necessary for confirming that mitigation has been effective in reducing road-kills, as management actions need time for animals to adapt to them (e.g. Corlatti, Hackländer, & Frey-Roos, 2009; Soanes et al., 2013). This monitoring

should be focused on population-level impacts and include fieldwork and genetic analyses (Corlatti et al., 2009).

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DATA AVAILABILITY STATEMENT

Data can be downloaded in most of the cases from those databases indicated in Appendix S1. In some cases, data cannot be shared because they have been obtained upon agreement from data providers. However, in these latter cases, researchers can get in contact with the persons listed in the Appendix S1 to ask permission to access these data. Raster layers of the model can be accessed at: <https://datadryad.org/stash/share/A5idNBDRVp6YHGg-i-x6aySAL19NcBs1sG-TLx8KvaQ>.

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BIOSKETCH

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section.

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