Do well-connected landscapes promote road-related mortality?

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ORIGINAL PAPER

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Abstract Cost surface (CS) models have emerged as a useful tool to examine the interactions between landscapes patterns and wildlife at large-scale extents. This approach is particularly relevant to guide conservation planning for species that show vulnerability to road networks in human-dominated landscapes. In this study, we measured the functional connectivity of the landscape in southern Portugal and examined how it may be related to stone marten road mortality risk. We addressed three questions: (1) How different levels of landscape connectivity influence stone marten occurrence in montado patches? (2) Is there any relation between montado patches connectivity and stone marten road mortality risk? (3) If so, which road-related features might be responsible for the species' high road mortality? We developed a series of connectivity models using CS scenarios with different resistance values given to each vegetation cover type to reflect different resistance to species movement. Our models showed that the likelihood of occurrence of stone marten decreased with distance to source areas, meaning continuous montado. Open areas and riparian areas within open area matrices entailed increased costs. We found higher stone marten mortality on roads in well-connected areas. Road sinuosity was an important factor influencing the mortality in those areas. This result challenges the way that connectivity and its relation to mortality has been generally regarded. Clearly, landscape connectivity and road-related mortality are not independent.

Keywords Carnivores · Stone marten · Habitat fragmentation · Hierarchical partitioning · Montado · Roadkill

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Introduction

Much research has shown that species respond strongly to the different spatial patterns created by different kinds of landscape change (e.g., Bissonette 1997, 2007; Hames et al. 2001; Lindenmayer and Fischer 2006; LaRue and Nielsen 2008), and responses may not be in accordance with human visual perception of the landscape pattern (Bissonette 2003). The main process that is associated with landscape change is habitat fragmentation, with loss being the primary driver of biodiversity decline (Fahrig and Merriam 1985; Pimm and Askins 1995). Because connectivity, among other ecological functions, facilitates biological fluxes such as animal dispersal and gene flow, mitigating the effects of habitat fragmentation and loss has become a key conservation issue (Hale et al. 2001; Moilanen and Nieminen 2002; Goodwin and Fahrig 2002). Landscape connectivity, the degree to which the landscape facilitates or impedes the movement of organisms among patches (Taylor et al. 1993), depends not only on the spatial pattern of the

landscape elements (structural connectivity) but also on the interactions between pattern and the biological characteristics of the species (functional connectivity), such as their ability to use and move across different landscape elements and their risk of mortality (Uezu et al. 2005; FitzGibbon et al. 2007). For example, a rodent and an ungulate occupying the same landscape will have different functional connectivity based on body size, speed of movement, and other life history characteristics.

Hence, connectivity, the connectedness between patches of suitable habitat for an individual species (Lindenmayer and Fischer 2006), is also a growing concern in roaded landscapes (Iuell et al. 2003). In fact, even in structurally connected areas, roadkill, noise, road surface and car avoidance (the avoidance effect), road size, and traffic volume (Jaeger et al. 2005) may affect suitable areas that become increasingly more fragmented (Forman et al. 2003). Therefore, landscape connectivity assessments are best accomplished if associated with road network characteristics (e.g., Eigenbrod et al. 2008). Nevertheless, despite the well-documented cumulative effects of landscape change and road mortality on biodiversity conservation, their effects often are evaluated independently. Landscape connectivity assessments generally take into account the amount and spatial arrangement of suitable habitat patches, but often exclude anthropogenic features (Mortelliti and Boitani 2008). Similarly, the effects of roads may be evaluated using mortality risk or road permeability, but often lack a landscape context (Ramp et al. 2005). If the objective is a more accurate assessment of landscape constraints on species movement and mortality risk, we argue that studies that account for the effects of both variables are more meaningful.

Sunquist and Sunquist (2001), Crooks (2002), and McRae et al. (2005) have shown that some endangered mammalian carnivores are particularly vulnerable to habitat fragmentation and road network expansion and are facing threats of local extinction in some regions (Ferreras 2001; Naves et al. 2003). Although the stone marten is currently a widespread and common carnivore, recent research documented that they are particularly sensitive to forest fragmentation in cork oak woodlands, hereafter called montado (Virgós and García 2002; Santos-Reis et al. 2004; Mortelliti and Boitani 2008), and are vulnerable to traffic (Grilo et al. 2009). They are the second most frequently killed carnivore on roads in Portugal during the period when females are provisioning young (Grilo et al. 2009).

In this study, we evaluated the functional connectivity of the montado ecosystem for stone marten in southern Portugal. The montado is an agroforestry–pastoral ecosystem that consists of scattered tree cover (60–100 trees per hectare) dominated by cork, *Quercus suber*, and holm, *Quercus rotundifolia*, oaks with interspersed pastures and agricultural fields (clover, wheat, barley, oats) (Pereira and Pires da Fonseca 2003). We were interested in examining the influence of differing functional connectivity on road mortality risk. We addressed three questions: (1) How do different levels of landscape connectivity influence marten occurrences in montado patches? (2) Is there any relation between landscape connectivity and stone marten road mortality risk? (3) If so, are there road-related features that might be responsible for the species' high road mortality? We developed a series of connectivity models using cost surface scenarios (CSS). In each CSS, we assigned different values to each vegetation cover type (Ray et al. 2002) to reflect different resistances to stone marten movement. In this way, each CSS represented a landscape with different permeability. We then examined how changing resistance values for cover types influenced the model fit of stone marten patch occurrence. We used the best fit model to evaluate what role connected areas played in the mortality of stone marten. We expected that as montado connectivity increased, the likelihood of stone marten presence in montado patches would also increase. High landscape connectivity is partially the result of the presence of low resistance montado patches and riparian areas. We expected that landscape connectivity would be negatively correlated with road mortality. Although the likelihood of stone marten presence is higher in connected montado areas, we predicted that inter-patch movements would increase the likelihood of encountering a road (Ferreras 2001). The large number of alternative crossing structures coupled with relatively low traffic volumes suggests a low mortality risk, but road sinuosity with its resultant shorter visibility sight lines suggests an increased risk of road-related mortality.

Material and methods

Study area

The study area covers 7,680 km^2 in the Alentejo province of southern Portugal (Fig. 1a). This region is characterized by its vast plains, with elevations ranging from 200 to 500 m.a.s.l. The climate is Mediterranean, with mild winters and hot and dry summers. Mean annual temperature is 15.6°C and mean annual precipitation 500 mm. The landscape is dominated by a cropland matrix including pastures and meadows (hereafter called open areas, 44%), montado (39%), followed by conifer and eucalyptus plantations (8%), and fruit orchards and olive groves (7%). Excluding the two major cities of Évora and Beja, with a mean density of 1,980 inhabitants per square kilometer, the remaining area has 21 inhabitants per square

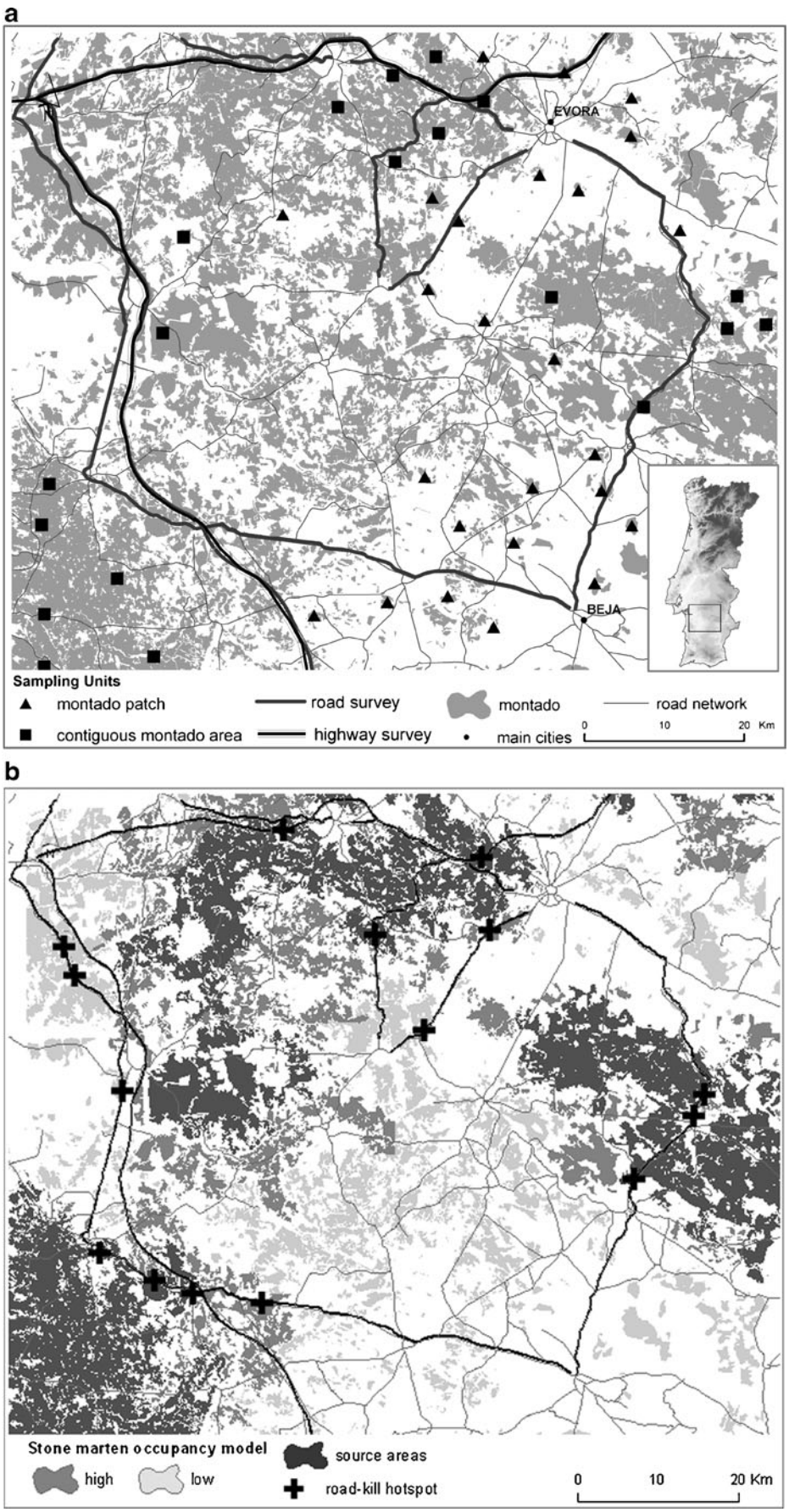

Fig. 1 **a** Study area in southern Portugal with the location of the sampling units to assess stone marten patch occupancy. **b** Stone marten occurrence likelihood (low<75%, *light gray*; high>75%, *medium gray*; source areas, *dark gray*) and roadkill hot spot locations (*plus sign*). *Dark lines* are the surveyed roads

kilometer (source: Instituto Nacional de Estatística 2002). Mean road density was 0.27 km km^{-2} and included approx. 160 km (8%) of highways. Traffic volumes ranged between 6,000 and 13,000 vehicles per day.

Land use map

Prior to field work, we created a land use map using geographic information system (GIS) software. We used georeferenced SPOT imagery (1998), with a spatial resolution of 20 m, to update COS'90 land cover units (a vectorial information produced by photo interpretation of 1990s aerial photographs; source: Instituto Geográfico Português) through screen digitization. We assessed the accuracy of the final land use map in the field. We found <5% differences between the cover types from the map and our verification in the field. We then aggregated the land cover classifications from the updated land cover into four land use categories which, according to the literature, influenced stone marten movements (e.g., Santos-Reis et al. 2004; Matos et al. 2008). These included: montado (MNT); open areas (OPEN); riparian areas (RIP); and OTHER uses, including conifer and eucalyptus plantations, orchards and olive groves, and urban areas. Then we converted the land use map in a grid using a cell size of 200 m (a trade-off between what is biologically meaningful for the species and computationally feasible). Any use of trade names is for descriptive purposes only and does not imply endorsement by the US Government.

Stone marten survey

From May to September 2005, we surveyed 44 sampling units (SU); 25 were located in smaller sized montado patches, ranging from 55 to 526 ha in size, and 19 in continuous montado areas, 11,668–60,175 ha (Fig. 1a). Sampling units were located at least 4 km apart to prevent sampling repeated individuals (mean distance=6,838± 2,249 m). We proportionally sampled each SU by placing from 7 to 17 scent stations in a 300-m interval grid; the number of scent stations depended on the extent of the site. A scent station consisted of one plate (1×1 m) covered with marble dust and was baited with chicken parts and lure (Cavens GUSTO Lure; Sargeant et al. 2003). Scent stations were checked for stone marten sand tracks every second day for a period of 6 days. For each SU, results were coded as either present (if detected at least once) or absent.

Stone marten roadkills were obtained from August 2003 to December 2007 on two types of roads: national roads and highways (Fig. 1a). National roads were one-lane in each direction with a low annual average daily traffic ranging from 500 to 2,161 vehicles/night (10 P.M.–6 A.M.) and a speed limit of 90/100 km h^{-1}. Highways were four-lane with a 7-m-wide median strip with an average of 330–2,494 vehicles/night and a speed limit of 120 km h^{-1}. We conducted roadkill surveys at low speed (approx. 50 km h^{-1}) along 6 segments of national roads (302 km) every 2 weeks (n=106 surveys), and mortality locations were recorded using a handheld GPS (maximum error 5 m). Roadkill data from highways (166 km) were obtained from the BRISA (private highway concession) database and locations were recorded daily by BRISA staff using the nearest 100 m (maximum error 50 m).

Data analysis

Landscape connectivity models

A series of landscape connectivity models (LCM) were developed using CSS with different resistance values to represent different movement costs (Ray et al. 2002; Adriaensen et al. 2003; Gonzales and Gergel 2007). The models were based on four assumptions: (1) Occupancy of smaller and more isolated montado patches is influenced by the effective distance to larger montado source areas, and this effect will become evident over time; (2) effective distances are dependent on different land use patterns with different resistance values; (3) roads are not likely to serve as barriers to movement; stone marten are known to cross roads (Grilo et al. 2009) and travel along streets in urban areas (Herr et al. 2009); and (4) water bodies have a negligible effect on carnivore movements due to their small size and limited number in the study area. We therefore assumed that different land cover types influenced stone marten movements and chose not to include roads or water bodies as landscape elements. We considered source areas as those sampling units with an occupancy probability ψ (the proportion of sites that species was detected) higher than 90%. To estimate ψ, we used a maximum likelihood approach that uses the repeated surveying of the sites following the MacKenzie et al. (2002) procedure in Program PRESENCE, version 2.3 (Hines 2006) at http://www.mbr-pwrc.usgs.gov/software/doc/presence/presence.html. We measured the cost to reach surveyed patches from the source areas using a set of eight CSS. These CSS were built by changing the relative resistance values of each land cover category. The resistance value in each cell in the land use grid was calculated as the distance to the source multiplied by the resistance value. We took into account different movement resistance scores related to MNT and OPEN. To examine the role of the riparian areas in different matrices, we developed different movement resistance scores for riparian areas within montado (RIP_MNT) and for riparian areas within open areas (RIP_OPEN). The different resistance weights for each CSS are given in

Table 1. For example, CSS A gave equal resistances for all vegetation cover types with a weight value=1; CSS B, C, and D show increasing resistance for OPEN and RIP_OPEN with resistance values of 2, 5, and 10, respectively, and with no effect for the presence of MNT and riparian areas RIP_MTN. Cost surface scenarios E, F, G, and H show low resistance for riparian areas RIP_MNT and RIP_OPEN with a resistance value = 1) and with varying resistance values for MNT (1 or 2) and OPEN (5 or 10). Mean cost resistance values of all CSS were then assigned to each of the 44 sampling units. The cost distance analyses were conducted with tools from the Spatial Analyst and Patch Analyst extensions (http://flash.lakeheadu.ca/~rrempel/patch/) of ArcView 3.2 (ESRI 1999).

We used a generalized linear model (GLM) logistic regression to identify whether CSS fit the data on stone marten presence and absence at each SU patch. We log-transformed all variables to achieve normality assumptions (Quinn and Keough 2002). We ranked models by Akaike's information criterion (AIC) score and AIC differences (ΔAIC) were determined. We considered that ΔAIC < 2 indicated a good fitting model (Burnham and Anderson 2002). We also assessed model fit using the Nagelkerke R^2 statistic, a measure analogous to the R^2 of least squares estimated regression models (Nagelkerke 1991). We also used the receiver operating characteristic (ROC, Deleo 1993). We evaluated model accuracy through the use of a ROC plot which graphs the proportion of true positives (sensitivity) in relation to the proportion of false positives (specificity) for a range of threshold values. The area under the ROC curve (AUC) is a reliable measure of overall model accuracy and commonly varies from 0.5 (random classification) to 1 (perfect discrimination). AUC values over 0.7 indicate good model performance (Fielding and Bell 1997). We also applied the K-fold cross-validation prediction error for generalized linear models to test the accuracy of the best model (Manly et al. 2002). This function calculates the mean sum square of the differences between the observations and the predicted value (Davison and Hinkley 1997).

Table 1 Cost surface scenarios with values of relative connectivity for land uses: MNT, OPEN, RIP MNT, and RIP OPEN (A–H)

LCM	MNT	OPEN	RIP MNT	RIP OPEN
A	1	1	1	1
B	1	2	1	2
C	1	5	1	5
D	1	10	1	10
E	2	5	1	1
F	1	5	1	1
G	2	10	1	1
H	1	10	1	1

Resistance values ranged from 1 to 10, with 10 being the greatest resistance possible for a landscape cover type

Roadkill analyses

We used the linear nearest neighbor analysis (Levine 2004) to evaluate whether stone marten casualties were clustered or dispersed along road segments. The linear nearest neighbor index (NNI) is expressed as the ratio of the observed mean distance divided by the expected mean distance (Clark and Evans 1954). The expected mean distance is the average distance between neighbors in a hypothetical random distribution. The mean nearest neighbor distance is given by the equation:

$$\mathrm{MeanNN}_{dist} = \sum_{i=1}^{N} \frac{\mathrm{NN}_{dist}}{N}$$

where N is the number of roadkills. The mean random distance is defined as

$$\mathrm{MeanRD} = 0.5\sqrt{\frac{S_{Total}}{N}}$$

where S_{total} is the total length of the surveyed road. If the observed mean distance is the same as the mean random distance, then the ratio will be 1. If the observed distance is shorter than the random, the NNI is <1 (clustered). The locations are dispersed when NNI is larger than 1.

Roadkill hotspots were defined for each 1,000-m road segment, equivalent to the mean radius of a stone marten home range (Santos-Reis et al. 2004) according to the procedure of Malo et al. (2004). We compared the observed number of collisions in each 1,000-m segment with the number of collisions expected in a random situation. If random, the likelihood of collision would show a Poisson distribution; we can identify a hot spot or cluster of stone marten mortalities on the road by the minimum number of collisions in a road segment that is extremely unlikely (<5%) to occur. By using nearest neighbor hierarchical clustering (Levine 2004), we identified the spatial location of these hot spots. We placed buffers with a 1,000-m radius around each hot spot centroid. We selected the same number of buffers in road segments without road kills (reference comparisons) with the Hawth's Analysis Tools extension ArcGis 9.2 (ESRI 2006) and compared the average stone marten likelihood of presence (estimated by the best GLM) model between hot spots and non-hot spots with a Mann–Whitney test (Dytham 2003). If we found significant differences between hot spots and reference comparisons, we then identified road-related variables that could be related to mortality risk. Seven road-related variables that might influence the animal–vehicle collisions

Table 2 Road-related variables used in roadkill hot spots analyses

Variables	Definition	Source	Type	Range
Road type	1—national road 2—highway	IgeoE	Nominal	1 or 2
Permeability_n	No. of passages	Field work	Scale	0–9
Permeability_wt	Sum of passages width (overpasses, underpasses, culverts, viaducts)	Field work	Scale	0–142
Sinuosity_c	No. of curves	IgeoE	Scale	0–7
Sinuosity_index	Total length of the road/distance between the start and finish locations	IgeoE	Scale	1.00–1.12
Fences	Presence/absence of fences	Field work	Nominal	0;1
Traffic	Daily traffic volume at night in 2005 (no. of vehicles/night[c])	IEP	Scale	351–2,494

IgeoE Instituto Geográfico do Exército, *IEP* Instituto de Estradas de Portugal

[a] 10 P.M.–6 A.M.

were estimated for each hot spot and non-hot spot buffer within the same connectivity landscape level using GIS (Table 2). We used GLM logistic regression to identify the road-related features influencing the presence/absence of stone marten hot spots. We used variance partitioning procedures (Chevan and Sutherland 1991; Mac Nally 2000) to assess the independent and joint effect of each road-related variable on hot spots buffers. This protocol employs goodness-of-fit measures for all 2^N possible models for N predictors instead of identifying a single best model (Mac Nally 2002). Variances are partitioned so that the total independent (I's) and joint (J's) contributions of explanatory variables can be estimated. This procedure allowed the identification of variables likely to have the greatest influence in explaining the locality of mortality hot spots. We used log-likelihood as a goodness-of-fit measure. To assess the statistical significance of variables, the data matrix was subjected to 1,000 randomizations to compute I's distributions for each predictor. Following Mac Nally (2002), if the observed value of I_{obs} for each variable was extreme (>95 percentile) relative to the generated distribution, then we retained the variable as potentially important for explaining hot spot occurrence. Results are expressed as Z scores and the statistical significance based on the upper 95% confidence limit ($Z > 1.96$; Mac Nally 2002). All statistical analyses were performed using the R statistical software, version 2.9.1 (R Development Core Team 2006, Vienna, Austria). GLM, K-fold cross-validation, and hierarchical partitioning were fitted using the *glm*, *boot*, and *hier.part* packages, respectively.

Results

Landscape connectivity models

We detected the presence of stone marten in 13 out of 25 smaller montado patches (52%) and in 18 continuous montado areas (95%). The similarity between observed occupancy (N) and estimated probability of occupancy (ψ) (N=0.52 and ψ=0.56 for open montado patches and N=0.95 and ψ=0.95 for continuous areas) demonstrated that our sampling method detected stone marten presence quite well. The difference in probability of stone marten presence between continuous areas and patches supported the interpretation that continuous areas can be regarded as source areas.

Four least cost models with similar AIC values fit the data quite well (Table 3), suggesting that they are reasonably equivalent. Overall results show that stone marten presence was positively related to effective shorter distances to source areas. Both open areas and riparian areas within open areas contributed to a lower connectivity for movements. Moreover, the least cost distance performed better than did Euclidian distance (CSS A) in describing stone marten presence. The best model was cost surface scenario D (β=−1.0529, SE=0.6495; Table 3): It suggests that stone marten presence decreased as the distance to source areas increased and indicates that open areas and riparian areas within open areas tend to be particularly

Table 3 Summary of parameters estimated for LCM

Model	AICc	ΔAICc	w	R^2	AUC
D	36.5224	0.0000	0.222	0.03	0.7435
C	37.4069	0.8845	0.142	0.06	0.7051
H	37.9666	1.4442	0.108	0.12	0.6923
F	38.3645	1.8422	0.088	0.16	0.6666
B	38.5722	2.0498	0.079	0.04	0.6602
G	38.6491	2.1267	0.076	0.07	0.6666
E	39.0288	2.5064	0.063	0.06	0.6153
A	39.1401	2.6177	0.059	0.09	0.5897
Null	37.1628	0.6400	0.161	–	–

AICc Akaike's information criteria adjusted for small sample size, Δ*AICc* difference between AICc of each model and the minimum AIC found for all the models compared, w probability that each model is the true model, R^2 Nagelkerke R^2, *AUC* area under the ROC curve

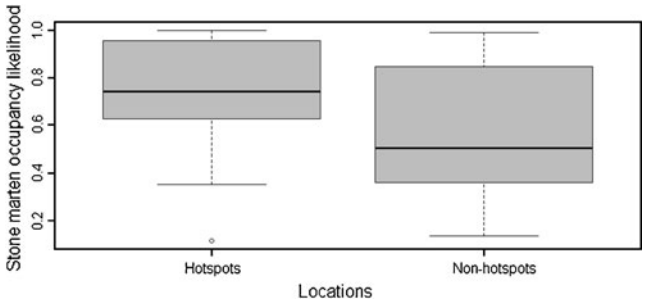

Fig. 2 Box plots for the hot spot and non-hot spot locations in relation to the likelihood of stone marten montado occurrence

costly. The model correctly classified presences (69%) and absences (66%). The AUC was 0.74 and appeared to be a good predictor for the presence of stone marten. The Nagelkerke R^2 was 0.16, suggesting that an important part of stone marten presence is explained by landscape connectivity. The prediction error estimated by cross-validation was 0.26, which means that the model fits the data well. If we can extrapolate the model to the entire study area, 76% of the montado in the study area is likely (≥75% probability) to be occupied by stone marten (Fig. 1b).

Hot spot analyses

We documented 182 stone marten fatalities on 468 km of national roads and highways in Alentejo. Although not significant, the nearest neighbor index suggested that stone marten casualties were clustered (0.94). Following Malo et al. (2004), we defined as hot spots road sections of 1,000-m length that had at least four road kills over the study period as hot spots. We identified 15 roadkill hot spots (Fig. 1b) and randomly selected another 15 sites without road kills as reference comparisons. Significant differences were found between hot spots and reference comparisons for stone marten presence likelihood derived from best the LCM model (Mann–Whitney test: U=63.5, n_0=15, n_1=15, p<0.05). On average, we found a 73% likelihood of stone marten presence in hot spot buffers and 51% likelihood in reference site buffers (Fig. 2). In order to identify the road-related variables that influenced the incidence of mortality in well-connected areas (average 0.79 stone marten patch occupancy likelihood), we chose other 15 non-hot spot buffered sites and ran variance partitioning analyses. Our analyses suggested that road sinuosity (Fig. 3) was the most important factor influencing the hot spot occurrences in well-connected areas, and it appears to be the important one for management purposes.

Discussion

Our study emphasized the value of distance from continuous forested areas on stone marten patch occupancy and the importance of roads as a disruptive factor in the directionality of their movement. We found one unexpected outcome that may change the way that connectivity is conventionally viewed. Although some studies have suggested that inter-patch movements may increase with habitat fragmentation and therefore may also increase road-related mortality risk (Ferreras 2001; Adriaensen et al. 2003), we did not find a higher frequency of road casualties in more fragmented areas. Importantly, the incidence of roadkills was significantly higher in road segments traversing continuously forested habitat. It appears then that not only does a well-connected landscape promote movement but that it may also promote higher road-related mortality. Hence, both landscape connectivity and road-related mortality are intertwined. Extended road networks in well-connected landscapes appear to be a serious threat to long-term population stability and viability (Dixon et al. 2006) of many species. Additionally, road segments with higher sinuosity had higher road-related mortality. On highly sinuous road sections, both vehicle drivers and animals have little time to be aware of each other's proximity, resulting in a shorter response time to

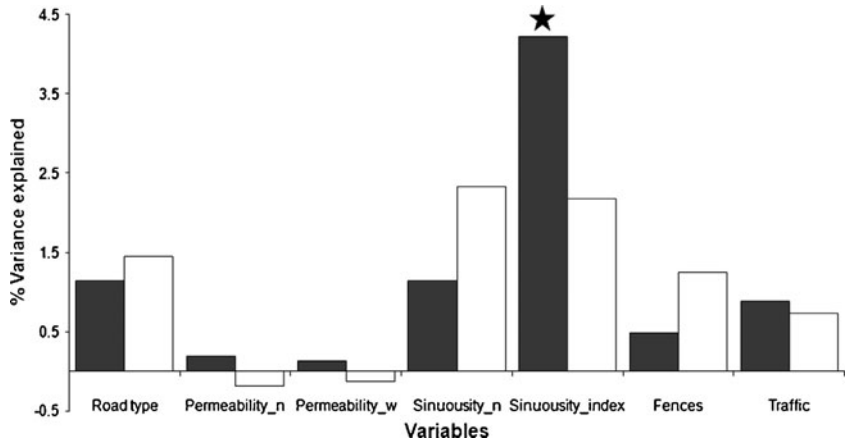

Fig. 3 Variance partitioning of the proportions of explained variance of road-related variables influence on roadkill hot spots. *Black bars* show the variance exclusively explained by the particular variable (I's); *open bars* show the variance explained jointly by the respective variable and the remaining variables. *Star* indicates significant result (being outside the 0.95 confidence range) after 1,000 randomizations

avoid collision. Grilo et al. (2009) reported similar results for genet and Egyptian mongoose (*Herpestes ichneumon*). In connected areas, cutting or removal of the vegetation along road verges of highly sinuous road sections may reduce suitability for the species, both in terms of cover and prey, which might prevent accidents by improving the visibility.

Evaluation of patch resistance relative to species movement characteristics can provide important insights to better understand landscape permeability and its implications for road-related mortality. Cost surface analysis is an important and useful tool that can help guide conservation planning in landscapes changed by humans and assist in the design of landscape management strategies at broader scale extents (Adriaensen et al. 2003; LaRue and Nielsen 2008). Clearly, patch occurrence is species-specific and influenced by the nature of the matrix (With et al. 1997; Tischendorf and Fahrig 2000; D'Eon et al. 2002). In fact, mobility is intimately related with species capabilities for moving across landscape features with different competition and predation exposure, energy costs, and availability of food and shelter to individuals (Johnson and Gaines 1985), and will affect access to and occurrence of patches. The vulnerability of species to fragmentation appears to depend heavily on matrix quality and its associated regional context, an often unappreciated relationship. The inclusion of the matrix characteristics and the non-Euclidian distance in the models strongly improved our understanding of the structural connectivity of the habitat. Our experience corroborates results from a growing number of studies (e.g., Hunter et al. 2003; Larue and Nielsen 2008).

Stone marten distribution patterns suggest that open areas act as semi-permeable filters, slowing or circumventing movement. Their avoidance of open areas is well documented (Rondinini and Boitani 2002; Virgós and García 2002; Goszczyński et al. 2007; Mortelliti and Boitani 2008), but how far they will move across open areas is not well understood. Additionally, riparian areas within open areas appear to be of little importance as potential corridors in promoting connectivity among the montado patches. This is in contrast to what has been recently reported, especially for carnivores in fragmented Mediterranean ecosystems (e.g., Virgós 2001; Matos et al. 2008). Rondinini and Boitani (2002) emphasized the importance of a wide variety of fruits and plants, as well the high density of rodents in the riparian areas, as important items in stone marten's diet (Genovesi et al. 1996), particularly in the dry season when their availability is high in the surrounding matrix, even if the matrix is montado (Matos et al. 2008). In fact, the riparian areas found in our study area are in poor condition, with only a few meters' width of shrubs (Matos et al. 2008), which may hamper carnivore mobility. Hence, riparian habitat restoration initiatives within unsuitable areas could enhance their role as potential corridors. Forest patch size was highly correlated with stone marten occurrence and is widely reported as one of the most important features influencing species occupancy (Michalski and Peres 2005), particularly with carnivores (Crooks 2002; Virgós et al. 2002). However, the presence of stone marten in smaller forested patches suggested that individuals moved across the area to supplement resources (Dunning et al. 1992; Rödel and Stubbe 2006). Similar movement also has been reported by Palomares and Delibes (1994) for genets *Genetta genetta* in a suboptimal fragmented habitat in Northern Doñana National Park (Spain).

In this study, we argued that the cumulative effects of landscape fragmentation and road mortality on biodiversity conservation are best understood if their effects are considered together. The opportunity exists in human-transformed landscapes to prevent further losses of carnivore species, but this will require additional focused research and conservation attention toward species, as well as incorporating adequate mitigation measures in future land use and road construction. The challenge remains to explore which processes underlie the complex relationship between mobility, spatial patterns, and riparian areas. Examination of possible distance threshold affects would appear to be important and should inform the potential for long-term population viability. We argue that the large-scale perspective presented here can prove useful in guiding local conservation efforts. We suggest that additional refinements are needed to examine site-specific issues at a scale extent appropriate for site planning (see Jepsen et al. 2005; Chetkiewicz et al. 2006). For example, a finer scale assessment of movement patterns of adults and dispersers can provide insights to assess the relative resistance of different vegetation cover types and assess the viability of a population at long term (e.g., Hunter et al. 2003; Broquet et al. 2006; Walker et al. 2007).

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