RESEARCH ARTICLE

Can *Eucalyptus* plantations influence the distribution range of mesocarnivores?

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Abstract

Context The expansion of exotic plantations can impose conservation challenges on wildlife, and the Iberian Peninsula has one of the widest planted areas of exotic *Eucalyptus* sp. in Europe. Since mesocarnivores are pivotal elements of ecosystems' functioning and *Eucalyptus* have been modifying the Portuguese landscape context in the last half century, it is crucial to understand how these systems may affect carnivores' range.

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/s10980-023-01787-8.

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Objectives We aim to identify the drivers of five mesocarnivores' distribution in Portugal (e.g., land-cover, ecogeographic predictors, mammal prey availability) and understand the influence of *Eucalyptus* plantations in their distribution range.

Methods Using generalized linear models, we modelled the distribution range of mesocarnivores. The initial dataset was randomly split for model training and validation, and the multicollinearity between the predictors was tested. Then, we examined the potential relationship between the *Eucalyptus* plantations area and the predicted probability presence of each species.

Results We detected species-specific patterns explained by different drivers, including climatic, land cover and mammal prey related ones. Furthermore, in areas of *Eucalyptus* plantations, the probability of occurrence of most Portuguese mesocarnivores is lower: red fox, stone marten, European badger, and Egyptian mongoose.

Conclusions Managers must take action to adapt their management to promote native forest patches within plantation, and allow the development of some understory within stands, to improve this plantation's permeability to mesocarnivores. This will increase the spatial heterogeneity and enhance resource availability, reducing the constraints that plantations might have on the range of mesocarnivores in Portugal.

Introduction

The Mediterranean Basin is a crossroad of species of African and European origins (Critical Ecosystem Partnership Fund 2017), which together with a historical human presence that shaped the region's environmental conditions, resulted in one of the most important biodiversity hotspots (Myers et al. 2000; Critical Ecosystem Partnership Fund 2017). Within the Basin, the Iberian Peninsula is a pivotal wildlife conservation area, due to the interaction between biodiversity values and historical landscapes composed, for instance, by *Dehesas* (Spanish name) or *Montados* (Portuguese name) (Maranon 1988). But in many areas the increasing human demand for wood and paper pulp has led to the replacement of the native (e.g., shrublands) and some of those historical habitats (e.g., woodlands) by plantations, especially *Eucalyptus* sp. (Deus et al. 2018; DGT 2020; FAO 2020).

The introduction of *Eucalyptus* in the Iberian Peninsula dates to the late 19th century and the area covered has been expanding ever since, with an increasing rate since the 1960s (FAO 1981; Alves et al. 2007; Veiras and Soto 2011; Deus et al. 2018). This exotic species, originally from Australia, can pose a threat to the conservation of wildlife species that have long been co-existing with, and adapting to, the traditional ecosystems of Iberia, especially if occurring as homogeneous and intensive plantations (da Silva et al. 2019). Various studies have highlighted the impacts of these plantations on local communities, inducing lower vertebrate diversity and abundance, often linked to limited food resources that can be used by these species (Rosalino and Santos-Reis 2009; Zahn et al. 2010; Calviño-Cancela et al. 2012; Martin et al. 2012; Teixeira et al. 2017; da Silva et al. 2019), or less shelter and protective cover (Teixeira et al. 2017; da Silva et al. 2019). Although this pattern is common in many areas where *Eucalyptus* has been established as a forestry species, there are only a few studies in Iberia, and particularly in Portugal, that have targeted the effect of *Eucalyptus* on the carnivore communities (but see Cruz et al. 2015; Teixeira

et al. 2023), some of which are species-specific (e.g., Castro et al. 2022).

Assessing how the implementation of *Eucalyptus* plantations can affect the carnivores' guild is pivotal for contributing to the maintenance of Iberia's natural heritage. Carnivores are core ecosystem elements that maintain biome functionality (Mangas et al. 2008; Roemer et al. 2009), landscape structure, and resilience (Roemer et al. 2009), due to, among others, their role as seed dispersal (Rosalino et al. 2010) and prey density controllers, i.e., small mammals (Salo et al. 2010; Williams et al. 2018). In altered landscapes, large or apex predators are often absent (Teixeira et al. 2020) as they are more sensitive to human pressure, disturbance, and habitat fragmentation and loss (Laliberte and Ripple 2004; Prugh et al. 2009). Inversely, mesocarnivores—mid-sized carnivore species with less than 15 kg (Roemer et al. 2009)—are much more diverse in their behavior and ecology, being often generalists and more resilient than apex species, reaching higher species richness and abundance in man-shaped environments when compared to large carnivores (Roemer et al. 2009). Owing to their smaller size and the ability to prosper in distinct habitats by using different resources, they often use landscapes that are shaped by humans and their activities (Alexandre et al. 2020). However, many of them are also less frequently the target of scientific studies in anthropic landscapes, such as plantations (but see Cruz et al. 2015; Bencatel et al. 2018), which constrain our ability to understand the effect of landscape changes on this group (see Márquez et al. 2022). Our ability to understand their ecological strategies to cope with the conservation challenges imposed by exotic plantations and man-related systems is therefore limited.

In Portugal, there are 15 species of carnivores, 13 of which are considered mesocarnivores (Álvares et al. 2019). Since not all mesocarnivores are widely distributed in mainland Portugal, we focused this study on the five terrestrial mesocarnivores with wider distribution ranges according to Álvares et al. (2019)—*Vulpes vulpes*—red fox; *Martes foina*—stone marten; *Genetta genetta*—common genet; *Meles meles*—European badger; and *Herpestes ichneumon*—Egyptian mongoose. Our main goal is to understand if the *Eucalyptus* plantations influence the distribution range of the most representative mesocarnivore community in continental Portugal.

Complementarily, we also aimed to identify other drivers (e.g., land cover, climate, mammal prey presence and/or richness, among others) that may affect the species distribution. We hypothesize that *Eucalyptus* plantations will globally constrain the distribution range of mesocarnivores in Portugal continental (Cruz et al. 2015) since these areas are usually poor providers of food and shelter for these species (Mangas et al. 2008; Ramírez and Simonetti 2011).

Methods

Study area

Portugal is the most Southwestern country of the Mediterranean Basin, with an area of 88,889 Km². The region's climatic conditions are characterized by humid cold winters and hot dry summers, and its landscape profoundly affects vegetation and wildlife (European Commission et al. 2010). Its geographical isolation and the great diversity of habitats and biotopes have allowed the development of its characteristic flora and fauna, including many endemic taxa (Myers et al. 2000; Pascual et al. 2011). The most representative land covers in the country are forests, accounting for 39% of the Portuguese territory, including native forests (e.g., *Quercus* deciduous forest, pine forest), high conservation value agroforestry systems (e.g., "montado" forests of *Quercus suber* and *Q. ilex*; DGT 2020) and exotic plantations. Of the entire forested area of Portugal, *Eucalyptus* plantations account for 27% (DGT 2020) (Fig. 1). From 2015 to 2018, the area of plantations of this exotic tree increased by 83,000
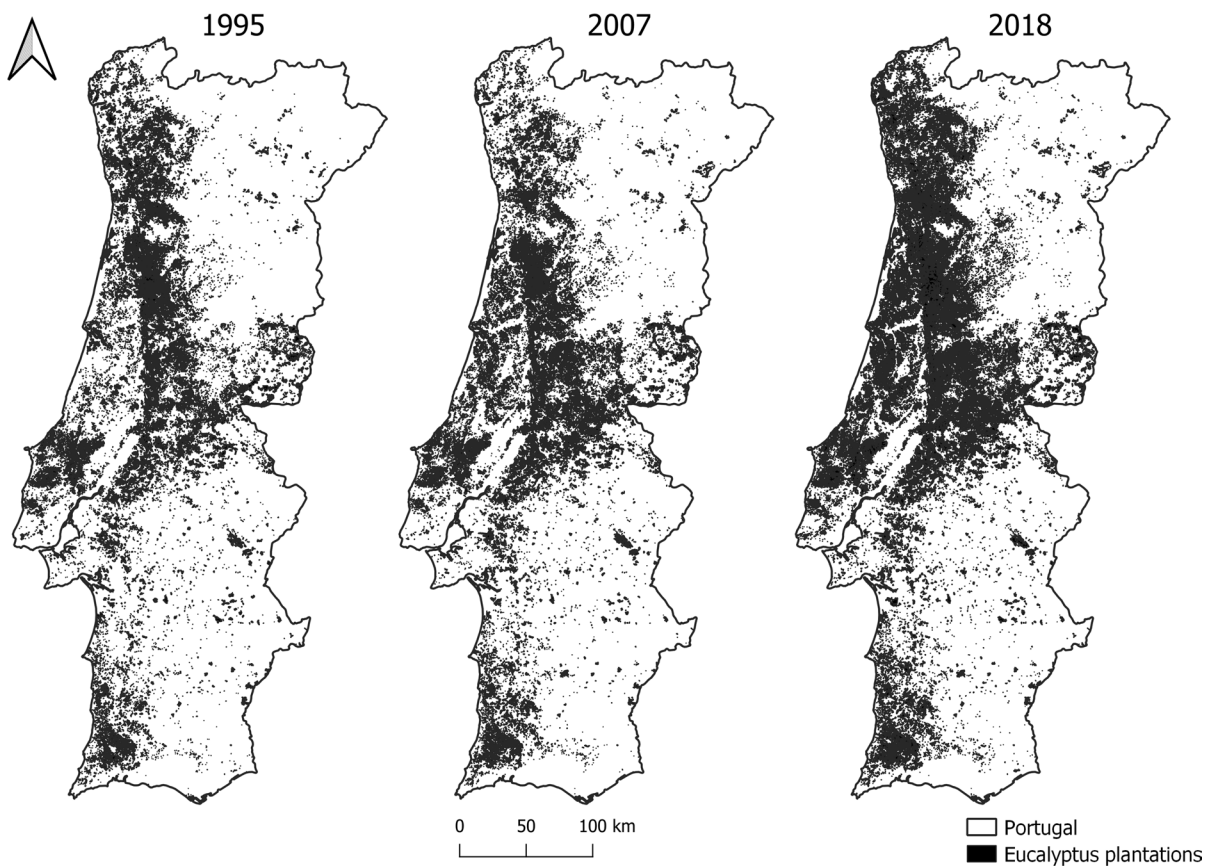
Fig. 1 *Eucalyptus* plantation evolution since 1995, 2007, and until 2018 using data collected from Land Use and Occupancy Map of Continental Portugal (COS 2018) available for download at https://snig.dgterritorio.gov.pt/rndg/srv/eng/catalog.search#/metadata/b498e89c-1093-4793-ad22-63516062891b

hectares, totalizing, currently, 928,000 hectares nationwide (ICNF 2015; DGT 2020).

Sampling scale and data collection

Continental Portugal was divided into a UTM grid of 10 × 10 km (N = 1005 territorial units). We then extracted the presence records of five mesocarnivores (red fox, stone marten, common genet, European badger, and Egyptian mongoose) from the "Mammal Atlas of Portugal" (https://atlasmamiferosportugal.wordpress.com/; Bencatel et al. 2019). We only collected records labeled as 'confirmed', including all unequivocal records, such as direct observation or capture of live animals, dead animals, specimen photos, and genetically identified samples (Bencatel et al. 2019). The collected presence records of all species are dated until the end of 2018 (see Figure A.1—presence map for each species; Appendix A).

Drivers of carnivores' distribution

To identify the drivers of mesocarnivores' distribution range in continental Portugal, we calibrated spatially explicit species distribution models based on ecogeographical predictors (EGVs), namely climatic, topographic and land use-related variables (Rahbek et al. 2007). We tested 38 EGVs (climatic, land use, topographic, and anthropical variable) based on their potential capability to explain the occurrence of the species targeted in this study (Martínez et al. 2022; Torre et al. 2022) (see Table B.1 Appendix B). In addition, we estimated four prey species presence and richness-related variables [European rabbit (*Oryctolagus cuniculus*), Iberian hare (*Lepus granatensis*), rodent richness, and insectivorous mammals' richness] in each territorial unit of 10 × 10 km, using the data collected by Bencatel et al. (2019), as prey availability may constrain mesocarnivores occurrence (Foster et al. 2013; Vilella et al. 2020).

All the information collected for the different variables listed in Table B1; Appendix B, estimated for each 10 × 10 km cell, was processed using the software QGIS, version 3.10.8 (QGIS Development Team 2020).

Data analysis

The dataset was randomly split into 70–30% subsets for model training and validation, respectively. To avoid multicollinearity between the numerical predictors, we assessed the predictor's variance inflation factor (VIF). The VIFs were analyzed using the Heiberger method (Heiberger 2012) with R package 'usdm' (Naimi et al. 2014) to find a set of predictors without collinearity (i.e., VIF < 3; Zuur et al. 2010). All non-collinear variables (see Table 1) were finally used to parameterize Generalized Linear Models (GLMs) for explaining species distribution range (one model for each species), using binomial distribution and a logistic link function (Cameron and Trivedi 2013). All the continuous variables were standardized. The best models for each species were produced on the training datasets by using a forward-backward stepwise procedure, based on Akaike's information criteria (AIC) (Burnham and Anderson 2002). We calculated the difAIC, i.e., the difference between the AIC of each species null model and the AIC of produced models including drivers, to access the models' usefulness (i.e., when difAIC > 2). As GLMs are based in presence/absence data and data for carnivore presence can be incomplete due to the low detectability of this group of species, we also performed species distribution models using presence-background Maxent models (Phillips et al. 2006). We conducted a Spearman correlation analysis between the predicted values generated by the Maxent and Generalized Linear models, to assess the agreement of both analytical methods and thus the robustness of our results. Maxent results were summarized in Appendix D. As a strong concordance was obtained in the predicted patterns from GLMs and Maxent (Figure D.1 and Table D.1; Appendix D; strong correlations - > 0.8 - across all species), subsequent analyses were carried out only with the GLMs results.

We assessed the models' predictive performance on the validation datasets by estimating the area under the ROC curve (AUC). According to Manel et al. (2001), AUC values between 0.5 and 0.7 indicate low accuracy, while models with values between 0.7 and 0.9 can predict species presence accurately and AUC > 0.9 indicates that models have high accuracy. The AUC value was computed using the 'ROCR' R package (Sing et al. 2005). The reliability of the predicted probabilities was assessed by exploring the

Table 1 List of final ecogeographical variables with VIF < 3 used in the GLMs to assess the drivers of the mesocarnivores distribution range (for the complete list of predictors see Table B.1—Appendix B in the Supplementary Material)

Variable	Description
Bio 3	Isothermality
Bio 15	Precipitation seasonality
AnnualPET	Mean annual potential evapotranspiration (mm / year)
Slope	The slope is the change (in degrees) in elevation over a certain distance
X1_Territ	Anthropic land use cover (ha)
X2_Agric	Agricultural areas cover (ha)
X4_SAF	Agroforestry areas cover (ha)
X5_Conif	Coniferous forests cover (ha)
X5_Invasive	Invasive species forests (without *Eucalyptus*) cover (ha)
X5_Broadleaf	Broadleaf forests cover (ha)
X6_Scrub	Scrub cover (ha)
X7_OpenArea	Open spaces (or with little vegetation) cover (ha)
X9_WetLand	Wetlands and surface water cover (ha)
Riq_Roedor	Rodent richness
Riq_Insect	Insectivorous mammals' richness
Rabbit_p	European rabbit presence
Hare_p	Iberian hare presence

calibration plots based on validation datasets (Pearce and Ferrier 2000). We plotted the observed frequency of each species' presence against the predicted probability of presence, using the R package 'ggplot2' (Wickham 2016). Once the models were validated, they were projected to the full extent of continental Portugal, using the R package 'fuzzySim' (Barbosa 2015).

In this first phase of the analysis (Fig. 2), we excluded the *Eucalyptus* plantation cover area from the independent variable's set. Our main goal was to test if carnivores' range could be constrained by *Eucalyptus* cover, but the higher importance of other predictors as determinants of carnivore's range can mask the detection of the effect of plantations' cover (i.e., the spatial pattern of *Eucalyptus* could be explained by a combination of EGV that could dilute or even mask its potential relationship with the target species). Thus, we opted to first produce good-fitting models, based on the most commonly identified environmental drivers (described in Table 1), and in a second phase (Fig. 2), assess if there was a relation between the predicted probability of presence and the *Eucalyptus* proportion of cover. Thus, we performed a linear regression for each species' predicted presence probability and the proportion of area covered by *Eucalyptus* plantations in each 10 × 10 km square.

We assessed if *Eucalyptus* plantations cover predictor could account for some of the probability of species occurrence variation, by comparing the AIC of the null model (without predictors), with that of the model that included the *Eucalyptus* cover proportion. We estimated the difAIC, but this time we used the difference between the AIC of the null models and that of produced models including *Eucalyptus* cover, for each species, and considered that there is very high or high support for the potential relation of *Eucalyptus* land cover proportion in driving species distribution range when difAIC > 10 or difAIC > 6, respectively (Burnham and Anderson 2002). The statistical analyses were carried out in R 4.0.5 (R Core Team 2021).

Results

All selected species (red fox, stone marten, common genet, European badger and Egyptian mongoose), showed a presence record covering most of continental Portugal. Nevertheless, Egyptian mongoose and stone marten were not detected in Northwestern and in most of the coastline respectively (Figure A.1; Appendix A).

Fig. 2 The different phases of the statistical analysis carried out to disentangle the effect of *Eucalyptus* plantations on the distribution range of mesocarnivores in Portugal
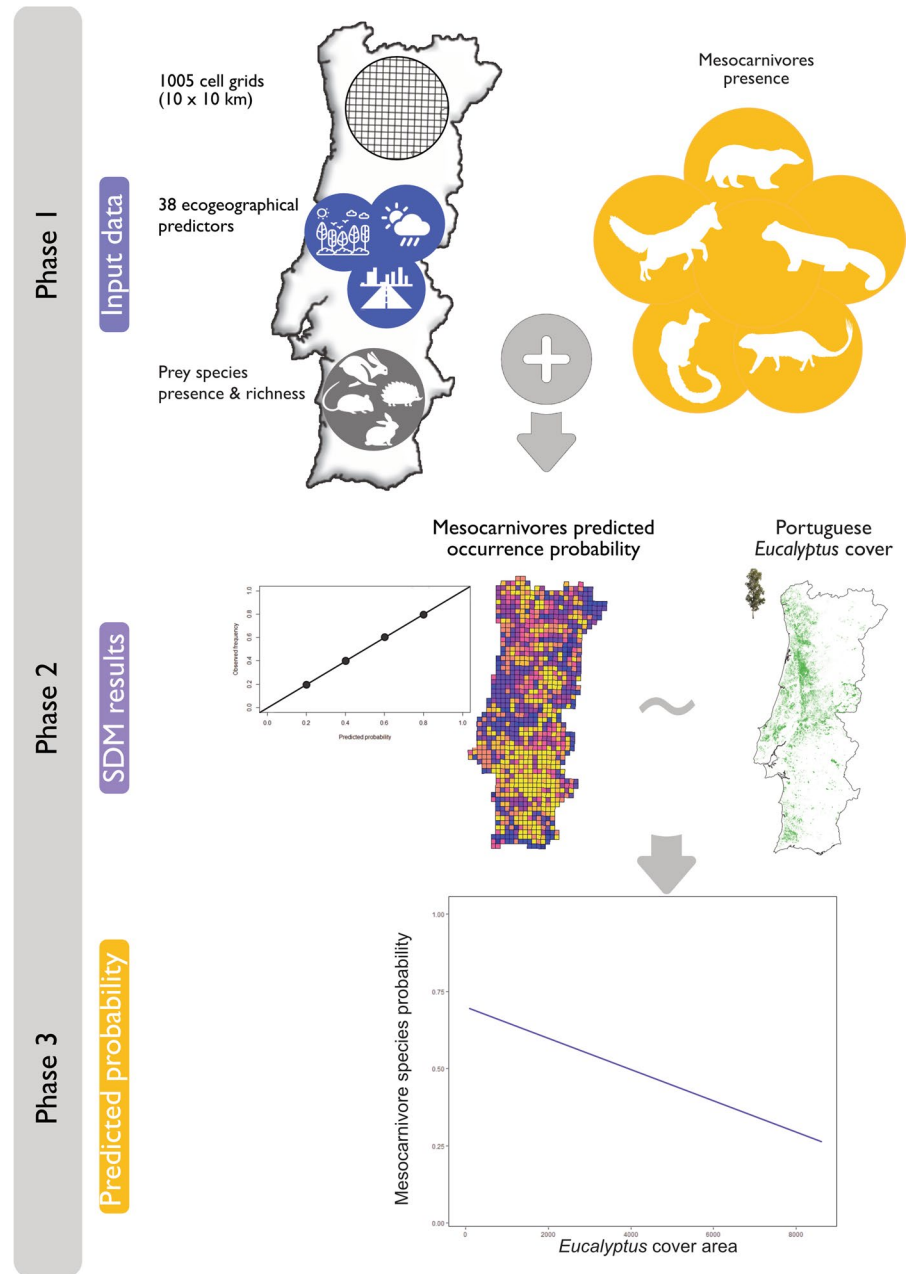
From the initial set of predictors, 25 were removed from the analysis due to collinearity problems (i.e., VIF > 3; see Table B.1; Appendix B). The remaining 13 EGVs and the four prey variables listed in Table 1, were used in the subsequent models. The most parsimonious model for each species is summarized in Table 2.

Globally, the presence of both lagomorphs (i.e., European rabbit and Iberian hare) has a positive influence on the mesocarnivores distribution and were the most influential variables in the majority of the analyzed species. However, there are some specific variations, with species presence probability being significantly higher in areas with higher slope (red fox and stone marten), mean annual potential evapotranspiration (annualPET; stone marten), precipitation seasonality ('Bio 15'; Egyptian mongoose), lower scrub ('X6_Scrub'; European badger and

Table 2 Best-fitting model for each targeted mesocarnivore species

Predictors	Coefficients	SE	p-value	CI 95%	
Red fox					
Intercept	− 0.002	0.105		− 0.207	0.204
Rabbit_p	1.645	0.246	***	1.180	2.147
Hare_p	1.386	0.250	***	0.909	1.893
Slope	0.472	0.120	***	0.242	0.714
X4_SAF	0.363	0.124	**	0.128	0.615
X6_Scrub	− 0.278	0.111	*	− 0.499	− 0.062
X2_Agric	0.163	0.095	.	− 0.021	0.352
X5_Conif	0.140	0.096	ns	− 0.046	0.331
Stone marten					
Intercept	− 1.240	0.141	***	− 1.522	− 0.969
Hare_p	1.093	0.209	***	0.685	1.505
AnnualPET	0.518	0.138	***	0.251	0.793
Slope	0.574	0.130	***	0.321	0.833
Bio 3	0.309	0.122	*	0.075	0.554
Rabbit_p	0.407	0.203	*	0.006	0.805
Riq_Roedor	− 0.086	0.043	*	− 0.172	− 0.003
X5_Conif	0.206	0.094	*	0.020	0.391
X4_SAF	0.168	0.108	ns	− 0.043	0.384
X6_Scrub	− 0.206	0.120	.	− 0.447	0.026
Bio 15	0.208	0.121	.	− 0.029	0.447
European badger					
Intercept	− 1.599	0.132	***	− 1.864	− 1.347
Rabbit_p	1.270	0.201	***	0.878	1.667
X1_Territ	− 0.672	0.161	***	− 1.008	− 0.378
X6_Scrub	− 0.380	0.099	***	− 0.579	− 0.190
Hare_p	0.920	0.206	***	0.516	1.326
Bio 3	0.313	0.114	**	0.098	0.545
X5_Broadleaf	0.145	0.082	.	− 0.023	0.309
X5_Invasive	− 0.160	0.097	.	− 0.352	0.029
Egyptian mongoose					
Intercept	0.682	0.112	***	0.466	0.906
Bio 15	1.244	0.131	***	0.997	1.509
X1_Territ	− 0.328	0.098	***	− 0.528	− 0.142
Hare_p	0.517	0.226	*	0.078	0.966
X7_OpenArea	− 0.230	0.100	*	− 0.425	− 0.031
Bio 3	0.225	0.100	*	0.026	0.420
Common genet					
Intercept	− 1.397	0.121	***	− 1.640	− 1.164
Rabbit_p	1.199	0.184	***	0.839	1.562
Hare_p	0.911	0.190	***	0.539	1.283
X6_Scrub	− 0.309	0.095	**	− 0.502	− 0.127
X5_Conif	0.147	0.086	.	− 0.022	0.315

The predictor variables whose 95% confidence intervals (CI) of the coefficients do not include 0 are highlighted with gray Codes are described in Table 2

SE standard error

p-value: 'ns' p < 1; '.' p < 0.1; '*'p < 0.05; '**'p < 0.01 and '***'p < 0.001

common genet), and anthropic land cover ('X1_Territ'; Egyptian mongoose and European badger). All best models' AUC values were > 0.7, indicating their capability to accurately predicted the presence of species (Table 3). The predicted occurrence probabilities for each mesocarnivore species are present in Fig. 3, and the calibration plots of the observed frequency of each species' presence against the predicted probability of presence are represented in Figure C.1 (Appendix C).

The AIC values obtained for the models produced for every species were lower than that of the null models. The difAIC was always > 30, indicating that those models had high support when compared to the null models (Table 3).

The surface of *Eucalyptus* plantations within which 10 × 10 km squares was negatively related to four mesocarnivores' probability occurrence (Table 4). Only for common genets this pattern was not detected (Table 4).

Discussion

Influence of *Eucalyptus* plantations on mesocarnivores distribution

Eucalyptus plantations have been established in several European countries, but Portugal and Spain concentrate most of the European *Eucalyptus* plantations (Tomé et al. 2021). It is assumed that the high landscape proportion covered by these exotic plantations in Iberia would induce changes in the native
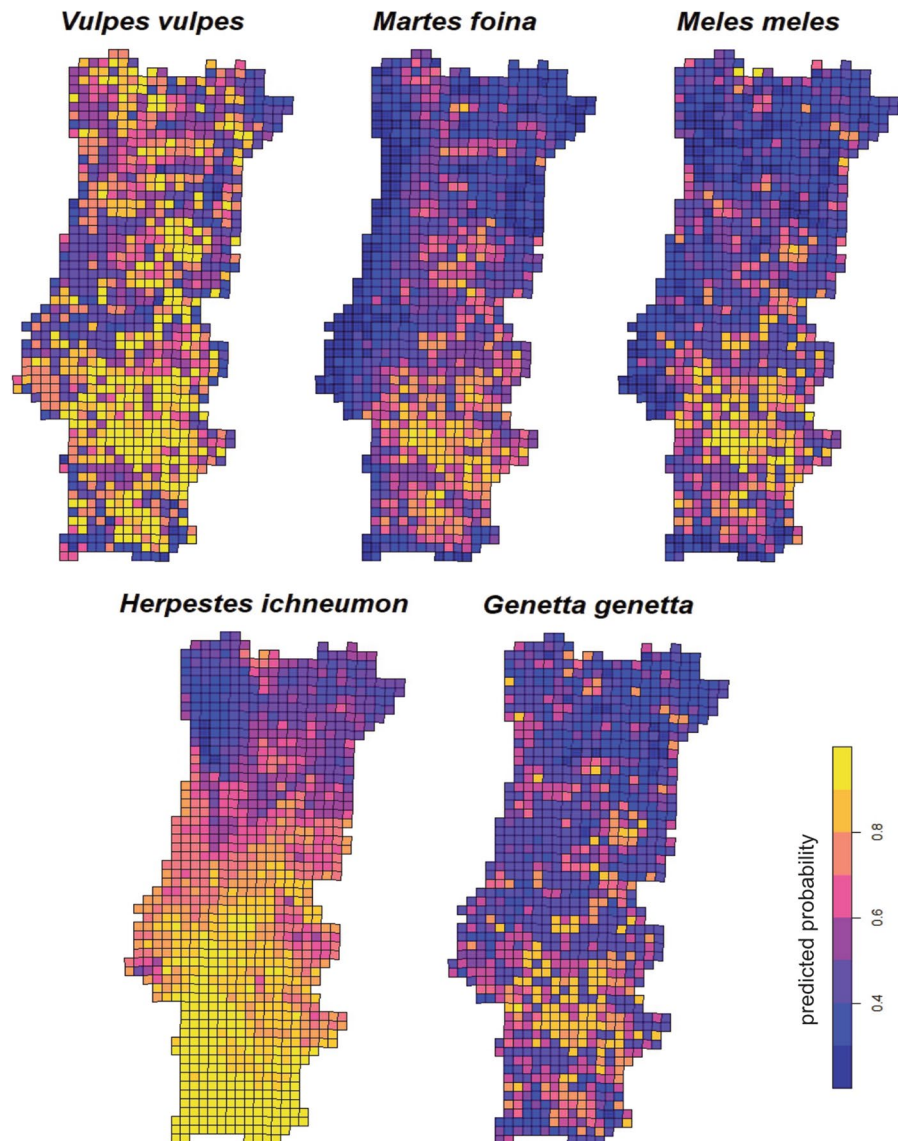
Table 3 Explained deviance and Akaike Information Criterion (AIC) values of the final models explaining the species distribution range in continental Portugal

Species	Explained deviance	AIC	difAIC	AUC
Red fox	17.0	763.23	138.71	0.740
Stone marten	15.9	768.48	120.88	0.755
European badger	17.8	746.50	144.31	0.762
Egyptian mongoose	19.7	746.58	170.52	0.786
Common genet	12.1	806.10	101.09	0.725

difAIC (i.e., the difference between the null models AIC of each species and the AIC of produced models including the selected drivers without *Eucalyptus* plantations cover area) and the Area Under the Curve (AUC) estimation for the best species distribution models produced for each species (Table 2)

communities, due to changes in resource availability, and several studies' results have corroborated this expected pattern (revised by Tomé et al. 2021). Nevertheless, most are of local/regional scope, and few targeted mesocarnivores (e.g., Cruz et al. 2015; Castro et al. 2022). Our national scale study on mesocarnivores fills the wide range study gap.

All mesocarnivores tested in our analyses seem to occur in distinct types of habitats, from anthropic landscapes, such as agroforest areas, to native forest habitats, emphasizing their ability to explore different habitats. Nevertheless, the presence of *Eucalyptus* plantations is consistently related to mesocarnivores distribution in Portugal, as areas with a higher cover of *Eucalyptus* plantations tend to have a lower presence probability by most species.

These constraints caused by the presence of plantations of this exotic tree can derive from the landscape transformation occurring when establishing plantations. Most *Eucalyptus* plantations, cultivated mainly for pulpwood production (Alves et al. 2018) are monospecific, harvested in 10–12 years rotation cycles (Alves et al. 2007; Silva and Tomé 2013), and show limited understory strata (da Silva et al. 2019). Such structure limits the type, diversity, and quantity of resources (e.g., prey species) they can provide for mesocarnivores (Carrilho et al. 2017; Teixeira et al. 2017), which may limit their use by these mammalian predators. But mesocarnivores' distribution in *Eucalyptus* plantations may not only be limited by the presence of prey (da Silva et al. 2019) but also by different factors such as refuge availability (e.g., lack of understory; Timo et al. 2014), species locomotion mode (e.g., arboreal species; Ferreira et al. 2018), disturbance scale (e.g., human activities; e.g., Castro et al. 2022), among others. All these factors depend on the type and intensity of management that will determine the stand structure (amount of understory available, presence of native trees inside stands that allows arboreal species to move to the canopy, etc.) and disturbance spatial and temporal scale (i.e., where, when and for how long will forestry workers be present within plantations and what kind of machinery will be used and when). Some authors have found a notorious preference for native and resource-rich habitats by badgers (Revilla et al. 2000; Rosalino et al. 2004, 2008; Cruz et al. 2015) as they select those habitat types due to key resources, such as food and shelter (Revilla et al. 2000; Rosalino et al.

Fig. 3 Predicted probability of mesocarnivores occurrence in continental Portugal at 10×10 km UTM squares level

Vulpes vulpes   *Martes foina*   *Meles meles*

Herpestes ichneumon   *Genetta genetta*

predicted probability

2004). Stone martens also prefer old-growth mixed woodland and evergreen oak forest, where food and refuges are more accessible, avoiding *Eucalyptus* plantations (Pereira et al. 2012; Cruz et al. 2015). This pattern is also observed at the macroecological scale of this study. Mesocarnivores' distribution ranges (Figure A.1 Appendix A) showed a clear lack of stone marten presence records along the entire Portuguese coastal zone, where there is a predominance of *Eucalyptus* plantations (Fig. 1). But this is probably not the only driver limiting stone martens' distribution, since the species is also absent in areas where *Eucalyptus* plantations are not present. Other

Mediterranean populations are absent or less common in areas with a high proportion of scrublands, croplands, and urban areas (e.g., Vergara et al. 2016). However, we were unable to detect a deleterious effect of these drivers on the Portuguese stone marten population. Therefore, other not accounted drivers (e.g., human density; Balestrieri et al. 2019) may be contributing to the detected pattern.

Regarding species locomotion mode, common genet has arboreal locomotion and isn't affected by *Eucalyptus* occupancy. Genets tend to prefer oak forests, often selecting *Quercus rotundifolia*, *Q. suber*, and *Arbutus unedo* woodlands (Zuberogoitia et al.

Table 4 Estimate, standard error (SE), *t*, and *p*-value, as well as the AIC value of the linear and null models produced to assess the effect of *Eucalyptus* plantations cover area on species distribution

Variable	Estimate	SE	*t* value	*p*-value	AIC model	AIC null model	difAIC
Red fox							
Intercept	0.661	0.007	100.891	***	− 303.546	− 298.043	5.504
*Eucalyptus*_cover	− 0.018	0.007	− 2.742	**			
Stone marten							
Intercept	0.323	0.006	50.348	***	− 345.958	− 338.400	7.558
*Eucalyptus*_cover	− 0.020	0.006	− 3.096	**			
European badger							
Intercept	0.328	0.007	47.112	***	− 183.430	− 178.138	5.292
*Eucalyptus*_cover	− 0.019	0.007	− 2.702	**			
Egyptian mongoose							
Intercept	0.642	0.007	87.928	***	− 83.742	− 74.447	9.295
*Eucalyptus*_cover	− 0.025	0.007	− 3.367	**			
Common genet							
Intercept	0.344	0.006	57.977	***	− 500.982	− 501.073	− 0.090
*Eucalyptus*_cover	− 0.008	0.006	− 1.381	–			

p-value: '.' p < 0.1; '*'p < 0.05; '**'p < 0.01 and '***'p < 0.001

All models presented in the table were significant with a p-value < 0.05

2002; Sarmento et al. 2010; Carvalho et al. 2014). They may avoid scrubland areas with specific composition (e.g., *Erica* spp. and *Cistus ladanifer*) and *Eucalyptus* stands (Sarmento et al. 2010). However, when *Eucalyptus* plantations include dense bramble understory (e.g., *Rubus* spp.) genets may still use these areas to avoid potential predators (e.g., dogs and cats; Zuberogoitia et al. 2002), as these patches provide an efficient refuge or competitors that may avoid plantations (e.g. foxes; Castro et al. 2022; Santos et al. 2007). This species also tends to reduce its presence in areas with frequent human disturbance (i.e., near houses and roads; Espírito-Santo et al. 2007).

Other drivers of mesocarnivores' distribution range

A myriad of other factors related to prey availability, distribution, climate, and land use contribute, isolated or in synergy, determine mesocarnivores' distribution in Portugal (Hipólito et al. 2018; Rosalino et al. 2019; Alexandre et al. 2020). Our data indicate that prey-related drivers can facilitate species presence in Portugal; concretely lagomorph presence, promote all five mesocarnivores' presence. European rabbit and Iberian hare are known as important food sources for many carnivores (Carvalho and Gomes 2001; Rosalino and Santos-Reis 2002; Sillero-Zubiri et al. 2004; Rosalino et al. 2009b; Verdade et al. 2011; Díaz-Ruiz et al. 2013). Therefore, their presence adds food resources to the landscape that can be used by

mesocarnivores, allowing them to inhabit regions where those prey subsist.

Inversely, human presence can restrain wildlife's distribution range (Oberosler et al. 2017), due to an increase in disturbance (including active persecution), changes in landscape structure, and available resources (Reason et al. 1993; Frick et al. 2020). Anthropic land uses (e.g., buildings, infrastructure, road networks, transportation, among others) can negatively influence some mesocarnivores (e.g., Egyptian mongoose and European badger) due to the increased disturbance they induce (Prigioni and Deflorian 2005; Barros et al. 2015). Predators may also minimize this disturbance effect by avoiding areas where humans are most common (e.g., plain or low slopes areas; Plate 2006), and using regions with higher slopes (e.g., in Canada, red foxes use the ravine slopes significantly more than expected since slopes were rarely used by humans; Adkins and Stott 1998). Two of the modelized species (red fox and stone marten) showed a preference for stepper areas, corroborating this avoidance behavior towards areas more used by humans.

Only two bioclimatic drivers, the precipitation seasonality, and the mean annual potential evapotranspiration, were influential in the distribution of species. Precipitation seasonality had a positive influence on Egyptian mongoose distribution. This species, whose core range is in Africa (Delibes 1999) is more likely to occur in areas that have a greater precipitation

variation between winter and summer, such as in southern Portugal, where summer is very dry (and the driest period is longer), and winter is wily wet, a typically Mediterranean climate. This effect is corroborated by the fact that there are practically no records of this species in the northwest of the country, a region with a typically Atlantic climate, where precipitation seasonality is lower (Álvares et al. 2019). The other important climatic driver, which affects stone marten, was evapotranspiration, which corresponds to the amount of water loss from evaporation as well as transpiration and is related to plant productivity. In areas with higher evapotranspiration, there is an increase in plant productivity. Therefore, food production in such regions is usually higher, promoting food availability for mesocarnivores (e.g., berries and fruit—important items in Mediterranean stone marten diet–Barrull et al. 2014; Lima 2021).

According to different studies, the landscape composition can be a pivotal driver of mesocarnivores' distribution. Scrublands can promote mesocarnivores' presence [e.g., genets (Virgós and Casanovas 1997), and badgers (Revilla et al. 2001)], by providing broader resource availability (e.g., food availability and shelter - Mangas et al. 2008; Carrilho et al. 2017). However, the influence of shrub cover depends on how well-developed and dense the understory is (Curveira-Santos et al. 2017), or the representativeness of another land cover, such as forest (Alexandre et al. 2020). For badgers, genets, and foxes, scrub areas had a negative influence on their distribution, although with a small effect (i.e., low coefficient values). We think that the lower food resources present in Mediterranean shrublands (when compared to other covers like riparian areas—Rosalino et al. 2009a) may overrule the shelter opportunities this land cover provides (Mangas et al. 2008).

Implications for *Eucalyptus* plantations management

Most species are usually more abundant, and the overall species richness is higher, in native forests when compared with plantations (Bremer and Farley 2010; Brockerhoff et al. 2013). But this overall pattern can vary, depending on the plantation's landscape structure and composition. In plantations where the canopies are not contiguous, a greater amount of light can reach the ground, thus promoting the development of the understory vegetation (i.e., herbaceous and shrub

layers), leading to higher prey abundances (Teixeira et al. 2017), which in turn will favor predators' presence (e.g., mesocarnivores). Some studies even show that *Eucalyptus* plantations with dense or complex understories can support species with densities and/or occurrence probabilities similar to those found in native forests (Fogarty and Vilella 2003; Silva-Rodríguez and Sieving 2012). These patterns show that management options can really make a difference in promoting regional biodiversity. Since natural areas are known to provide more resources and harbor a higher abundance of species (Torre et al. 2022), forestry managers need to prioritize the preservation of other habitats (e.g., riparian galleries) within *Eucalyptus* plantations, as they are key habitats to some carnivores (Gehring and Swihart 2003; Mestre et al. 2007; Matos et al. 2009). Patches of other habitats or linear structures (e.g., riparian forests, hedgerows, streams) within *Eucalyptus*, will allow the homogeneity of the plantations to be broken, add resources usable by wildlife, and can act as corridors for the species movements (Cruz et al. 2015), which altogether will facilitate mesocarnivore' presence and use of plantations, mitigating its possible barrier effect (Kupfer et al. 2006).

Conclusion

It is accepted that *Eucalyptus* plantations support lower levels of biodiversity than natural mixed forests (Brockerhoff et al. 2013), and several studies conducted in the Iberian Peninsula corroborated this pattern (see Tomé et al. 2021 for a summary of studies). Our study appears to follow similar reasoning by indicating that the distribution of most mesocarnivores species seems to be constrained by *Eucalyptus* plantations. However, it is not only the presence of this exotic plantation that influences mesocarnivores' distribution patterns. Other factors also contribute to the detected pattern, namely variables associated with prey availability, clime, other land use, and topography. Therefore, *Eucalyptus* managers need to consider the biodiversity dimension when defining their management plans and thus should adopt a strategy that minimizes the detected effect of plantations on mesocarnivores distribution, to assure that production is done in the most sustainable way possible and are not another biodiversity loss driver.

Data availability The datasets used during the current study are available at "Mammal Atlas of Portugal" (https://atlasmamiferosportugal.wordpress.com/; Bencatel et al. 2018).

Declarations

Competing interests The authors declare no competing interests.

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