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Progress in Identifying High Nature Value *Montados:* Impacts of Grazing on Hardwood Rangeland Biodiversity☆

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ABSTRACT

Due to their complex structure and traditional low-intensity management, Portuguese oak woodland rangelands known as *montados* are often considered high nature value (HNV) farming systems, and as such, they may be deemed eligible for subsidies and incentives by governmental and nongovernmental agencies. Too little is known about how the HNV concept might be applied to conserve complex silvopastoral systems. These systems, due to their structural and functional complexity at multiple scales, tend to support high levels of biodiversity. Montados are in sharp decline as a result of the rapid specialization of land management that, through simplification, undermines multifunctionality. Understanding how changes in management influence these systems and their biodiversity is needed for prioritizing conservation efforts and for ensuring they remain HNV systems. On the basis of a field survey in 58 plots distributed among 29 paddocks on 17 farms, we conducted an integrated analysis of the relationship between grazing intensity and biodiversity in montados of similar biophysical and structural characteristics. Data on management were obtained through interviews, and biodiversity data (vegetation, macrofungi, birds, herpetofauna) were obtained through specific field protocols. Additional spatial data, such as soil characteristics, slope, land cover, and linear landscape elements, were also analyzed. The results show no overall biodiversity variation as a result of different management practices. However, different groups of species react differently to specific management practices, and within a pasture, grazing impacts are heterogenous. In low grazing intensity plots, macrofungi species richness was found to be higher, while bird species richness was lower. Using tree regeneration as proxy for montado sustainability, results show less tree regeneration in areas with higher forage quality and more intense grazing. Pathways for future progress are proposed, including creating areas within a paddock that attract grazing away from where regeneration is desired.

Introduction

Silvopastoral systems cover several million hectares of the European Union's (EU) agricultural land. They are mostly found in the southern EU countries, where there are environmental conditions that limit intensive and specialized farming, such as poor and shallow soils, steep slopes, and scarce water, characteristics common on Mediterranean rangelands (Rigueiro-Rodríguez et al., 2009; Bergmeier et al., 2010; Plieninger et al., 2015). These land use systems combine forestry and livestock production, in systems using extensive livestock grazing practices and supporting varying tree densities (McAdam et al., 2009; Bergmeier et al., 2010; Plieninger et al., 2011). Because their heterogeneity and multiple vegetation layers create a diverse mosaic of habitats and support high levels of biodiversity (Bugalho et al., 2011; Godinho et al., 2011), these silvopastoral systems are often pointed out as outstanding examples of HNV farmland in Europe (Cooper et al., 2007; Paracchini et al., 2008; Oppermann et al., 2012). The HNV concept (Beaufoy et al., 1994) was introduced to the research and policy spheres in the beginning of the 1990s as a way to acknowledge the role of specific farming systems in maintaining biodiversity and landscape quality in the European countryside (Peneva et al., 2015; Strohbach et al., 2015; EFNCP, 2017). This concept is based on the assumption that low-intensity agricultural management leads to smaller production outputs but results in higher biological and landscape diversity levels on farmland (Doxa et al., 2010; Lomba et al., 2015). As a policy designation, it

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makes the land eligible for a number of incentive and subsidy programs in European countries and from the EU.

The emergence of the HNV concept accompanies an increased demand for environmental sustainability and countryside conservation (Oppermann et al., 2012). The European Union's Biodiversity Strategy focuses on sites with existing conservation status including sites designated as within the EU's *Natura 2000* program. Natura 2000 is an EU-wide network of core breeding and resting sites for rare and threatened species, as well as for some rare natural habitat types (Baker, 2003; Jones-Walters and Čivić, 2013; Kati et al., 2015). This does not address many places with conservation value that are at higher risk of land use and management change because they do not have conservation status. The HNV concept addresses such places, as it expands efforts to encourage environmentally positive farming systems and maintain biodiversity beyond Natura 2000 sites and other lands in conservation status.

The Common Agricultural Policy of the European Union (CAP), particularly the production programs that are its first pillar, is the most powerful instrument influencing land use decisions at the farm level in Europe (Latruffe et al., 2013; Renwick et al., 2013; Ribeiro et al., 2014). Despite its strategic goals, which include the conservation of natural resources and biodiversity (van Zanten et al., 2014), the CAP has led to two opposing trends affecting the welfare of silvopastoral systems (Pinto-Correia and Azeda, 2017). The first scenario is abandonment of grazing, due to new and complex rules that limit participation of rangelands with trees and shrubs in subsidy programs, causing landowners to convert these areas to either forestry or crop production rather than maintaining an integrated system (Trisorio and Povellato, 2010). In the second scenario, grazing is intensified in response to subsidies offered on a per head basis, resulting in higher livestock stocking rates. In the long run, overstocking reduces the productivity of pastures and inhibits tree regeneration (Pinto-Correia et al., 2014; Almeida et al., 2016; Ferraz-de-Oliveira et al., 2016; Godinho et al., 2016b; Guerra et al., 2016; Pinto-Correia and Azeda, 2017). The HNV of silvopastoral systems is not secured under either of these outcomes. Characterizing the biodiversity values of silvopastoral systems under different management regimes is key to focusing conservation efforts and targeting environmental compensation payments or subsidies in ways that effectively conserve European wooded rangelands.

The Portuguese *montado* is the silvo-pastoral system dominant in Southern Portugal and one of the most characteristic in Europe (Pinto-Correia et al., 2011b; Sá-Sousa, 2014). It covers approximately 1-m hectares and has a strong management tradition that integrates forestry and livestock production (Pinto-Correia and Godinho, 2013; Godinho et al., 2016b). The tree cover is composed of cork and holm oaks (*Quercus suber* and *Quercus rotundifolia,* respectively) at diverse densities (Godinho et al., 2016a), and the understory is either cultivated or in natural pasture, which may have shrubs, depending on management (Canteiro et al., 2011). Used for extensive livestock grazing, pastures may sometimes be cultivated to grow forage crops for supplementary animal feed. The low intensity of human use, together with different vegetation layers and the mosaic created by patches of differing tree and shrub densities, supports a diversified landscape and high levels of biodiversity (e.g., Martins-da-Silva et al., 2009; Bugalho et al., 2011; Godinho and Rabaça, 2011; Godinho et al., 2011; Almeida et al., 2013; Godinho et al., 2016a).

The montado is an outstanding illustration of the main challenges faced by silvopastoral landowners in the European context of changing and uncertain public policy interventions. Despite its conservation value, both market mechanisms and public policies (Almeida et al., 2013; Pinto-Correia and Azeda, 2017) create pressures for simplification of the system through specialization and intensification. The intensification of grazing explains ongoing changes in livestock grazing: replacement of sheep and Iberian pigs by cattle, replacement of indigenous cattle breeds by imported and heavier breeds, and higher stocking densities. These changes lead to soil compaction, increased fodder needs, and much stronger pressure on pastures and tree regeneration. With higher grazing pressure there are also fewer shrub patches. With less natural tree regeneration, tree cover tends to decline in density over time. The structural diversity of the montado is reduced, as well as its biodiversity value (Almeida et al., 2016; Pinto-Correia and Azeda, 2017). These trends mean that not all montados can still be considered HNV. But if those montados with HNV could be clearly identified, it would be possible to target public policies so that they would receive support for maintaining their conservation values.

The aim of this paper is to bring new insights into the assessment of the impact of management practices, particularly livestock grazing, on two aspects of montados that are important to maintaining a montado's classification as HNV: 1) biodiversity as represented by selected species groups and 2) tree regeneration and recruitment, as tree recruitment is crucial for a montado's structural diversity and a proxy for its stability.

We have applied an analytical approach integrating a set of different indicators of montado vegetation structure and biodiversity values and assessed how they are connected to management practices. Such an approach required disciplinary integration, both in the sampling design and for data collection protocols, as well as in the analysis. Social science and spatial analysis were particularly relevant to defining the sampling approach and integrating input from the different disciplines involved; social scientists designed the interview guide after consulting agronomy, forestry, and conservation specialists; the specialists for each taxonomic group were obliged to collaborate on the sampling strategy for each selected property so that data could later be analyzed jointly. The results are expected to bring useful insights informing the design of future policy tools for the montado.

Material and Methods

Study Area and Sampling Design

The study was conducted in the European Natura 2000 network site of Monfurado (PTCON0031), mainly located in the municipality of Montemor-o-Novo and partly in the municipality of Évora (Fig. 1), in southern Portugal. The site has ~24 000 ha, and the maximum elevation within the study area is 424 m above sea level. About 71.07% of this site is covered by well-preserved montados dominated by cork and holm oaks with a wide range of tree densities. This spatial fuzziness is a core characteristic of these silvopastoral land use systems (Pinto-Correia et al., 2011a; Godinho et al., 2016a), along with an understory mosaic of natural pastures, cereal and forage crops, and shrubs. In our study area, the management of the montado is mainly focused on livestock production, combined with the forest products of cork and wood for charcoal production.

In order to better assess management activities and variations, the study was focused on private montado properties managed for livestock production in combination with cork. Real-world situations are difficult to compare in detail due to their diverse characteristics. To minimize the variability associated with montado sample areas, the areas selected for sample plots were characterized by the following: a relatively dense tree canopy of 30 − 50% cover, dominance by cork oaks, similar soils with no registered limitations for grazing and forestry, and gentle slopes (0 − 15% slope).

We selected 58 sampling sites. One high-intensity and one low-intensity grazing plot of 1.5 ha were designated in each of 29 paddocks (average size: 42.93 ha; minimum size: 10.56 ha; maximum size: 97.35 ha). Paddocks were mainly grazed by cattle (12 parcels; mean stocking density = 0.629 LU/ha/yr) or jointly by cattle and sheep (9 parcels; mean stocking density = 0.667 LU/ha/yr). The remaining paddocks are grazed by cattle and pigs together (5 parcels; mean stocking density = 0.271 LU/ha/yr), pigs only (2 parcels; mean stocking density = 0.024 LU/ha/yr); and sheep only (1 parcel; stocking density = 0.292 LU/ha/yr).

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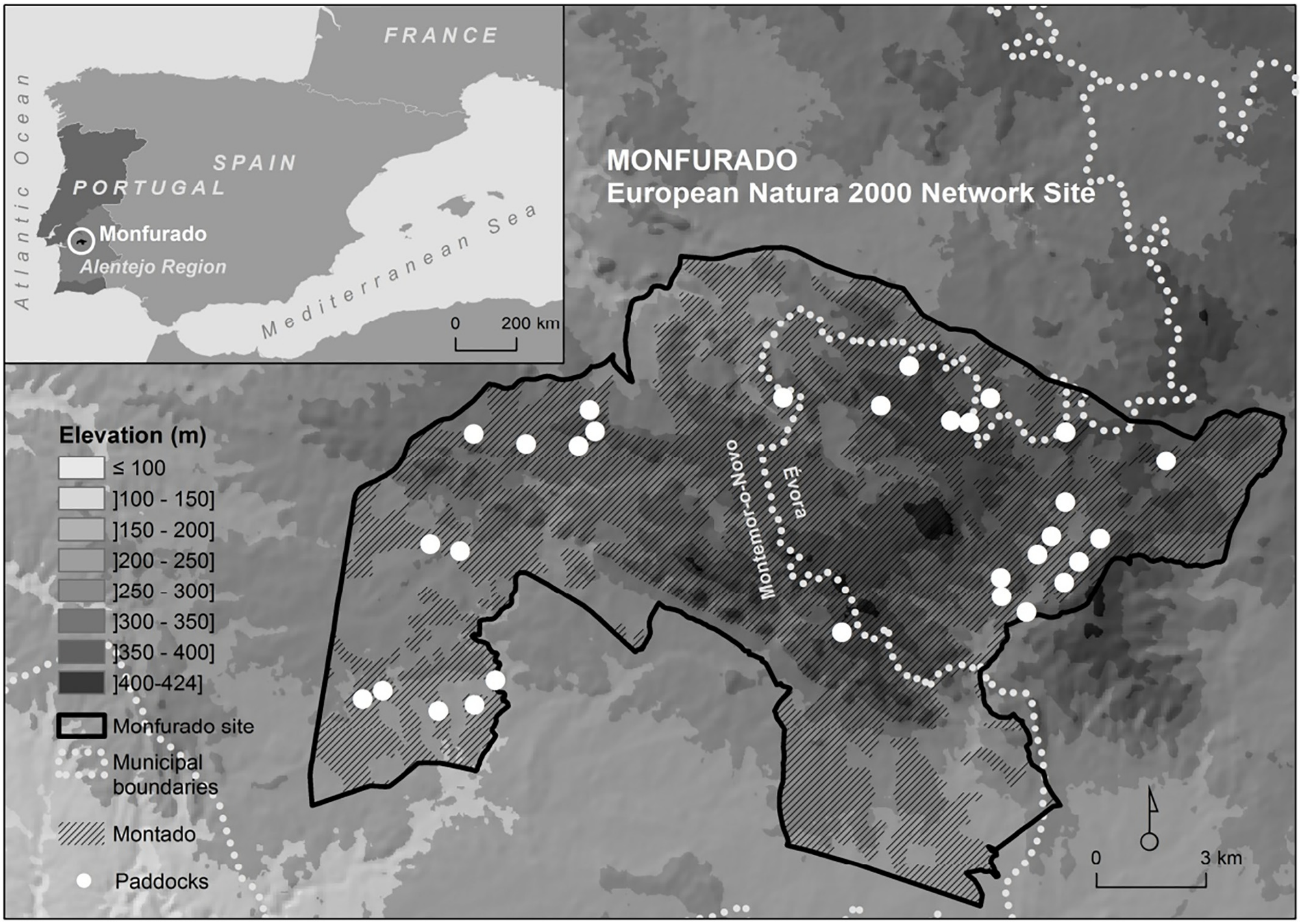

Figure 1. Location of study areas in the Monfurado Natura 2000 site.

Grazing intensity within each paddock is spatially heterogeneous. The location of the two sampling plots within each paddock was determined through two steps: 1) aerial photo visual appraisal for identification of watering and feeding points, as well as gates and resting locations, where grazing pressure is highest; and 2) based on these images, a face-to-face interview with the land manager to identify two sites with clearly contrasting levels of livestock grazing intensity, representing low- and high-intensity grazing.

Field Surveys and Data Collection

Two fundamentally different types of data were collected: 1) data on grazing management based on interviews with landowners or farm workers and 2) field data for different species groups to assess biodiversity impacts, including data on tree regeneration. When possible, the landowner was interviewed; otherwise, the interview was conducted with an employee indicated by the owner. The interviews took place on the property or at another location chosen by the interviewee, lasting from 1.5 to 3 hr and using the interview guide. Through the interviews, detailed data on the present management of the plots were obtained.

The semistructured interviews were divided into three sections: 1) property characteristics—size, number of paddocks, land cover, number and type of animals; 2) land use and management options—goals and management strategy of the owner, inputs and outputs from the property, management practices relating to soil conservation, shrub control, pasture improvement, management of the herd, animal feeding, tree pruning, tree cover regeneration, and detailed information on livestock management; and 3) management options and the drivers for decisions, including the role of the CAP and other policy tools. Focusing mainly on grazing management, the data collected included species, breeds, animal units, and permanence in each paddock. Livestock gates, watering, feeding, and resting locations were noted. During the field work (described in more detail later), dung density was also measured as a proxy for the spatial distribution of livestock. For the present analysis, the main outcome of the interview data was the identification of the two sampling plots within each paddock representing high and low grazing intensity, measured as a composite indicator of livestock type, breed, and days of residence on the plot.

For the analysis of biodiversity, different species groups were selected: macrofungal sporocarps, vegetation, reptiles, and birds. The field sampling approach followed a detailed plan using validated field protocols in order to obtain a standardized sampling structure for all species groups that was adapted to real-world situations (rangelands under private management). The fieldwork approach for the different species groups is described later.

Macrofungal Sporocarp Surveys

Macrofungal sporocarp surveys were conducted on the plots, along linear transects 20 m wide and 100 m long, during the winter months of 2013 and 2014. All macrofungal sporocarps found within transects were collected and recorded. Identification was carried out upon detection and whenever necessary specimens were kept in a freezer at 3°C for further verification. Each macrofungal taxon was placed in one of three main functional groups—saprotrophic, parasitic, or mycorrhizal—in agreement with Frade and Alfonso (2003), Kränzlin (2005), and Moreno et al. (1986).

Vegetation Surveys

Vegetation surveys were carried out in the plots during autumn through winter of 2013/2014 and in the spring of 2014. The sampling was defined according to the already validated protocols for this biophysical context (Mueller-Dombois and Ellenberg, 1974). Belt transects 20 m wide and of variable lengths sufficient to include 15 mature cork oak trees were used in vegetation sampling. The autumn-winter samplings focused on collecting data about the structure of cork oak stands and regeneration density, while spring sampling focused on floristics. These were used to calculate indicators of biodiversity. Floristic surveys were conducted in twenty 1-m^2 quadrats per plot (580 quadrats per grazing level) for plant cover, bare soil, species composition, and species abundance by visually estimating the vertical projection of plant canopies onto the ground, expressed as a percentage of quadrat area (Mueller-Dombois and Ellenberg, 1974). In order to capture differences in the structure of the vegetation in the tree layer, three indices were computed (adapted from DGF 2001 for our tree density classes):

$$\begin{aligned} TCI &= 1-\prod_{i=1}^{4}\left(1-\frac{C_i}{100}\right) \\ THI &= \frac{C_1\times 18 + C_2\times 5 + C_3\times 1.25 + C_4\times 0.25}{\sum_{i=1}^{4} C_i} \\ TDI &= \frac{\sum_{i=1}^{2} C_i - \sum_{i=3}^{4} C_i}{100} \end{aligned} \tag{1}$$

where TCI = Tree Cover Index, THI = Tree Height Index, TDI = Tree Dominance Index, and C_i the percent cover of each layer (C_4 – ≤ 0.5 m, C_3 – 0.5-2.0 m, C_2 – 2.0-8.0 m, C_1 – > 8.0 m).

Pasture samples were also collected to estimate the total biomass availability. Herbaceous material was collected at about 2 cm above ground level, using a cordless handheld grass cutter, from six quadrats (1 m^2) in each plot, randomly selected from those used for the vegetation surveys. Pasture samples were oven-dried (65°C, 72 hr) and weighed to determine biomass availability. Nutritive value, represented by protein (Dumas method; AOAC 990.03) and fiber (Van Soest method; ISO 16472: 2006) concentration, was determined in two composite samples obtained from the six initial pasture samples in each plot.

Reptile Sampling

Reptile sampling was conducted during the summer months of 2013 and 2014. The data were gathered combining 40- to 50-min zigzag walking transects with surveys in coarse woody debris and rock outcrops (e.g., Godinho et al., 2011), following the methodology described by Hill et al. (2005). Sampling was done during the daytime, with clear or partly cloudy sky and air temperature between 10°C and 25°C, when target species are more active.

Bird Censuses

Bird censuses were carried out using 10-min point counts (Bibby et al., 2005) with 25-, 50-, and 100-m radii, comprising a total area of 3.14 ha per sample point and allowing the determination of abundances. The surveys were conducted between sunrise and 10 a.m., when birds are more active. The first sampling period was in June 2013, and the second one in December 2013. All censuses have contributed to determining the indicators of bird diversity, and species were grouped in functional guilds, according to their habitat preferences, reflecting a complexity gradient of vegetation structure, from simpler habitats such as grasslands to oak forests (Catarino et al., 2016).

Additional Datasets

Data from the MODIS-based Vegetation Continuous Field (MOD44B) were used to extract the tree cover. We used data from 2014 as we did not find significant differences between 2013 and 2014. The MODIS-based Vegetation Indices (MOD13Q1) product was used to assess the variability in vegetation condition. The Enhanced Vegetation Index is more sensitive to structural variations in canopy, while the Normalized Vegetation Index is closely related to chlorophyll content (Huete et al., 2002). Because we intended to use these data as proxy variables for montado structure in each plot, we used images from the summer of 2014 because summer-season images provide a better spectral contrast between the overstory and understory in montado (Carreiras et al., 2006; Godinho et al., 2016a). The values of the vegetation indices were extracted using the centroid of each plot.

Data from the ASTER digital elevation model (DEM) were also used to compute surface metrics in order to assess the variability in more stable biophysical characteristics of montado landscapes, enabling an estimate of their influence on biodiversity variables. The original DEM, with 1 arc-second of spatial resolution, was resampled to a resolution of 30 m through the bilinear interpolation method in ArcGIS 10 (ESRI, 2011). Given the relation between landscape heterogeneity and biodiversity, indicators of surface complexity such as the terrain ruggedness index (Riley et al., 1999), surface relief ratio (Pike and Wilson, 1971), topographic position index (Guisan et al., 1999), and slope (Horn, 1981) were computed. In addition, radiation and moisture DEM-derived metrics were calculated because they can affect vegetation composition. These were the compound topographic index (Moore et al., 1993; Gessler et al., 1995), topographic wetness index (Schmidt and Persson, 2003), and topographic solar radiation aspect index (Roberts and Cooper, 1989). This set of surface metrics was obtained through Geomorphometry and Gradient Metrics (version a1.0) (Evans et al., 2013), DEM Surface Tools (Jenness, 2011), and Land Facet Corridor Tools (Jenness et al., 2013), toolboxes for ArcGIS 10 (ESRI, 2011). Mean values using a 3-cell window were extracted for each plot.

Data Analysis

The data collected, both from existing spatial information and from field work and farm interviews, have been grouped by the montado characteristics they represent (Table 1). These indicators express the diversity of data for the different taxonomic groups considered, as well as the biophysical and land cover characteristics of the plots.

To determine the relationships between biodiversity and grazing management in montados, due to the lack of similar analyses in the literature, an exploratory statistical approach was used. It was based on four main steps: 1) an exploratory data analysis based on nonparametric tests to search for significant differences in biodiversity indicators within low and high grazing intensity plots; 2) a principal component analysis (PCA) to identify associations among biodiversity indicators and also reduce the number of dependent variables used in the following step; 3) boosted regression tree (BRT), a machine learning algorithm capable of dealing with nonlinear responses and interactions among factors (Elith et al., 2008) to assess the factors behind the distribution of biodiversity indicators, using the PCA scores as dependent variables and the remaining variables listed in Table 1 (cork oak stand structure, biophysical characteristics, grazing intensity, and the main characteristics of the pastures in each plot) as explanatory variables; and 4) BRT was used to assess the factors explaining variation in cork regeneration.

Two nonparametric tests were applied: the Wilcoxon rank sum test and the Kruskal–Wallis test, to identify statistically significant differences between the biodiversity variables in plots classified as having high versus low grazing intensity (grazing intensity variable in Table 1). Box plots were used to represent the main findings. All the analyses were conducted in R version 3.1.3 (R Development Core Team, 2015).

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Table 1
Full set of indicators used to assess biodiversity distribution and cork oak regeneration in montado.

	Variable code	Description	Units
Biodiversity	MacF_SR	Species richness (macrofungi)	No. of species
	Sapro_SR	Species richness (saprotrophic fungi)	No. of species
	Myco_SR	Species richness (mycorrhizal fungi)	No. of species
	Rep_Pvi	Reptiles—*Podarcis virescens*	No. of observations
	Rep_Psa	Reptiles—*Psammodromus algirus*	No. of observations
	B_SR[1]	Species richness (birds)	No. of species
	AGB_SR[2]	Species richness (agricultural generalist birds)	No. of species
	FSB_SR[2]	Species richness (forest specialist birds)	No. of species
	FGB_SR[2]	Species richness (forest generalist birds)	No. of species
	BTS_SR[2]	Species richness (edge species—birds)	No. of species
	B_ShaDI[2]	Shannon's diversity index (birds)	–
	VEG_ShaDI	Shannon's diversity index (vegetation)	–
	VEG_SimEI	Simpson's evenness index (vegetation)	–
Cork oak stand structure	DBH	Diameter at breast height	cm
	TDens	Tree density	no./ha
	TCover[3]	Tree cover	%
	nTCover[3]	Non − tree cover (herbaceous and shrubs)	%
	BS[3]	Bare soil	%
	TCI	Tree cover index	–
	THI	Tree height index	–
	TDI	Tree dominance index	–
	SC[4]	Shrub cover	%
	SH[4]	Mean height of the shrub layer	cm
	NDVI[5]	Normalized difference vegetation index	–
	EVI[5]	Enhanced vegetation index	–
	DR1[6]	Seedlings: $h \leq 10$ cm	no./100 m^2
	DR2[6]	Saplings: $10 < h \leq 70$ cm	no./100 m^2
	DR3[6]	Saplings: $70 < h \leq 130$ cm	no./100 m^2
	DR4[6]	Young trees: $h > 130$ cm and DBH ≤ 9.5 cm	no./100 m^2
	DRt[6]	Sum of DR1-DR4	no./100 m^2
Biophysical metrics	SLOPE[7]	Slope angle	%
	TWI[7]	Topographic wetness index	–
	CTI[7]	Compound topographic index	–
	TRASP[7]	Topographic solar-radiation aspect index	–
	TRI[7]	Terrain ruggedness index	–
	SHAN[7]	Shannon diversity index	–
	SRR[7]	Surface relief ratio	–
	SPOSI[7]	Slope position	–
Grazing	GI	Grazing intensity	Binary (high or low)
	DD	Dung density	no./ha
	LD	Livestock density (at paddock scale)	LU[8]/ha/yr
	GII[9]	Grazing intensity index	–
Pastures	BA	Biomass availability	kg/ha (of DM[10])
	P	Crude protein	% DM[10]
	F	Fiber	% DM[10]

[1] Data from winter (w) and spring (s) surveys and total (t) species richness.
[2] Data from winter (w) and spring (s) surveys.
[3] Extracted from MODIS-based Vegetation Continuous Field (MOD44B).
[4] Data from autumn-winter (a) and spring (s) surveys and mean (m) values between both periods.
[5] Extracted from MODIS's MOD13Q1 product.
[6] DR: Density of regeneration; DRt: Total density of regeneration; h: height; DBH: Diameter at breast height.
[7] Methods used to compute DEM-derived metrics: slope (Horn, 1981); TWI (Beven and Kirkby, 1979; Schmidt and Persson, 2003); CTI (Moore et al., 1993; Gessler et al., 1995); TRASP (Roberts and Cooper, 1989); TRI (Riley et al., 1999); SHAN (Jenness et al., 2013); SRR (Pike and Wilson, 1971); and SPOSI (Guisan et al., 1999).
[8] Livestock units (1 LU is the grazing equivalent of one adult dairy cow producing 3 000 kg of milk annually).
[9] Scores from Principal Component Analysis (PCA) using GI, DD, and LD to reduce scalar limitations of GI and inspect relations between the previously mentioned variables. Factor loadings: DD: 0.88; LD: 0.88; GI: −0.01; Eigenvalue: 1.55; Proportion variance explained: 0.52. The "psych" package for R (Revelle, 2017) was used to compute the PCA.
[10] DM: Dry matter.

Given the high number of indicators, there was a need to reduce the complexity of the analysis while ensuring reliable results; we used the previous outputs, along with a correlation analysis between the biodiversity indicators, to identify highly correlated pairs (correlations > |0.7| following Tabachnick and Fidell's, 2007 suggestion), in order to reduce the number of biodiversity indicators. Only one variable from each group of highly correlated indicators was selected. The final set of variables was used in a PCA to identify relationships between the biodiversity indicators. In order to decide between orthogonal and oblique rotations, we followed the procedure suggested by Tabachnick and Fidell (2007). We have evaluated all the oblique rotations available in the "psych" package for R (Revelle, 2017) and inspected the correlation among factors. Since we did not find correlations above 0.32 among factors, we decided to use orthogonal rotations. The varimax rotation criterion (Kaiser, 1958) was used to explain the most significant gradients, simplifying the factors by maximizing the variance of the factor loadings (Tabachnick and Fidell, 2007). Uniqueness values were retained, representing the proportion of the variability of each variable not associated with the factors. Scores of the principal components with eigenvalues > 1 were retained and used as dependent variables in the subsequent analysis, to assess how different factors affect biodiversity distribution in the montado.

To identify the relations between biodiversity components and montado structure, biophysical factors, and grazing indicators, we applied BRT (Friedman et al., 2000) using the 'gbm.step' function and a Gaussian response type, with a learning rate of 0.001 and a bag fraction of 0.50. Tree complexity was set to 3, according to the suggestion of Elith et al. (2008) for small data sets. The relative importance of the predictor variables in the model delivered by BRT was retained (Friedman, 2001; Friedman and Meulman, 2003). To obtain more reliable results, we performed 100 runs and both mean and standard deviation values of the relative importance of the predictors are presented. For each biodiversity component, BRT models were simplified using the "gbm.simplify" function following a backward stepwise procedure (Elith et al., 2008). To better interpret the fitted functions, partial dependence plots were used to visualize the type of response of the predictors (Friedman, 2001).

The assessment of factors explaining the distribution of natural regeneration followed a procedure similar to that described previously for biodiversity. However, in addition to the indicators already described, we also used the variables indicating pasture quality as a proxy for animal distribution, following several authors who found positive relations between livestock distribution and forage use with forage quality (Roath and Krueger, 1982; Kie and Boroski, 1996; Ganskopp and Bohnert, 2009; Kaufmann et al., 2013a, 2013b).

Results

Factors Affecting Biodiversity in Montado

We collected and identified 145 species in the macrofungal sporocarp surveys (73 saprotrophic; 70 mycorrhizal; 2 parasitic). We identified 126 species in low-intensity grazing plots, while 90 species were identified in high-intensity grazing plots. In the vegetation surveys we identified 269 plant species. Of those, 172 were found in low-intensity grazing plots and 156 in high-intensity grazing plots. From the 12 reptiles identified during the surveys (9 in low-intensity grazing plots and 10 in high-intensity grazing plots), 6 lizards and 5 snakes were found, as well as the *Amphisbaenian Blanus cinereus,* a worm lizard.

Table 2
Results of the Wilcoxon rank-sum and Kruskal-Wallis tests performed to compare the biodiversity indicators in low- and high-intensity grazing plots.

	Wilcoxon rank-sum test		Kruskal-Wallis test	
	W	*P*	chi-squared	*P*
MacF_SR	**259.5**	***0.012***[1]	**6.304**	***0.012***[1]
Sapro_SR	370.5	*0.438*	0.613	*0.433*
Myco_SR	**260.0**	***0.012***[1]	**6.304**	***0.012***[1]
Rep_Pvi	513.5	*0.148*	2.107	*0.146*
Rep_Psa	363.0	*0.373*	0.806	*0.369*
B_SRw	**570.5**	***0.019***[1]	**5.530**	***0.018***[1]
AGB_SRw	**543.0**	***0.039***[1]	**4.292**	***0.038***[1]
FSB_SRw	459.0	*0.538*	0.388	*0.533*
FGB_SRw	500.5	*0.207*	1.609	*0.204*
BTS_SRw	513.5	*0.109*	2.593	*0.107*
B_ShaDIw	**550.5**	***0.043***[1]	**4.095**	***0.043***[1]
B_SRs	505.0	*0.185*	1.776	*0.182*
AGB_SRs	**579.0**	***0.010***[1]	**6.567**	***0.010***[1]
FSB_SRs	427.0	*0.920*	0.012	*0.913*
FGB_SRs	447.5	*0.674*	0.184	*0.668*
BTS_SRs	483.0	*0.284*	1.167	*0.280*
B_ShaDIs	460.0	*0.543*	0.378	*0.538*
B_SRt	**591.5**	***0.008***[2]	**7.164**	***0.007***[2]
B_ShaDIm	540.5	*0.062*	3.488	*0.061*
VEG_ShaDI	493.5	*0.259*	1.290	*0.256*
VEG_SimEI	409.5	*0.870*	0.029	*0.864*

MacF_SR indicates species richness of macrofungi; Sapro_SR, species richness of saprotrophic fungi; Myco_SR, species richness of mycorrhizal fungi; Rep_Pvi, number of observations of *Podarcis virescens;* Rep_Psa, number of observations of *Psammodromus algirus;* B_SR, species richness of birds; AGB_SR, species richness of agricultural generalist birds; FSB_SR, species richness of forest specialist birds; FGB_SR, species richness of forest generalist birds; BTS_SR, species richness of edge species—birds; B_ShaDIw, Shannon's diversity index of birds; VEG_ShaDI, Shannon's diversity index of vegetation communities; VEG_SimEI, Simpson's evenness index of vegetation communities. The results of the bird surveys are divided in winter (w), spring (s), and total species richness (t).

[1] $P < 0.05$.

[2] $P < 0.01$.

Concerning birds, 63 species were identified during the field surveys (46 species in the spring surveys—42 in low-intensity grazing plots and 37 in high-intensity grazing plots; 49 species in the winter surveys—32 in low-intensity grazing plots and 44 in high-intensity grazing plots).

On the basis of the results of the Wilcoxon rank-sum and Kruskal-Wallis tests (Table 2), a significant difference was observed between the distribution of species richness for macrofungi (also observed for mycorrhizal fungi) and birds (particularly for agricultural generalist birds and for total species richness) in low- and high-intensity grazing plots. However, the species groups responded differently to low- and high-intensity grazing (Fig. 2). While macrofungi are negatively associated with high-intensity grazing, a positive relationship is found between grazing intensity and birds.

After variable reduction, the PCA identified four uncorrelated linear combinations with eigenvalue > 1 (Table 3). The first one accounted for 22% of the total variation in the distribution of the biodiversity indicators, as this component positively related to the overall species richness of macrofungi. The second principal component accounted for 20% of variability in our plots. This component is positively related to the indicators of bird distribution. The third component identified in the PCA explained 17% of the existing variability in the diversity metrics. This principal component is positively related to vegetation diversity, particularly with Shannon's diversity index and Simpson's evenness index. The final component (accounting for 15% of the variation) is related to the distribution of the lizard *Podarcis virescens,* likely coinciding with areas containing the snags or decomposing dead trees and rocky outcrops characteristic of its habitat.

Concerning the effects of several indicator variables on the distribution of the four principal components obtained in the previous analysis, the results of the BRT models show differing influences (Fig. 3). While the distribution of macrofungi diversity (PC1) is strongly influenced by the percentage of bare soil, mean shrub height, terrain wetness index, terrain ruggedness index, and grazing intensity (Fig. 3a), the distribution of the bird diversity (PC2) is more influenced by shrub cover, the total cover index, and grazing intensity (Fig. 3b). In this latter case, and in contrast with the macrofungi, terrain wetness index and bare soil made rather small contributions to explaining the variability of bird diversity in montado plots. When considering diversity in vegetation (PC3), mean shrub height shows the greatest influence, followed by tree height index and shrub cover (Fig. 3c). Finally, an enhanced vegetation index, terrain ruggedness, terrain wetness index, percentage of shrubland and grassland cover, and bare soil are the indicators with the greatest importance for explaining variability in the field observations of *Podarcis virescens* (PC4; Fig. 3d).

The results of the BRT models after variable reduction through the *gbm.simplify* function, as well as the performance of the models, are presented in Figure 4 and Table 4, respectively. PC1 scores (mainly related to species richness of macrofungi) increase with increased values of bare soil between 10% and 20%, higher vegetation height, higher values for the terrain ruggedness index (less affected by livestock grazing), and lower grazing intensity (Fig. 4a). PC2 scores (essentially determined by bird diversity) are positively related to lower values for shrub cover, terrain ruggedness, and tree density and more livestock grazing (Fig. 4b). These results point out the high prevalence of generalist species in the sample plots, species less sensitive to disturbance. PC3 scores (closely related with vegetation diversity) are higher in areas with a high tree height index, increasing with the mean height of the shrub layer but decreasing with the percentage of non − tree cover layers (Fig. 4c). Contrary to what we have observed previously, areas with more grazing seem to have higher vegetation diversity. Although this is the model with the lowest performance, this higher diversity may be due to the greater dominance of ruderal therophytes in these areas. On the one hand, PC4 scores, representing the distribution of the number of observations of *Podarcis virescens,* increase with higher values for the terrain ruggedness index (between 3.5 and 4.0), the terrain wetness index, and

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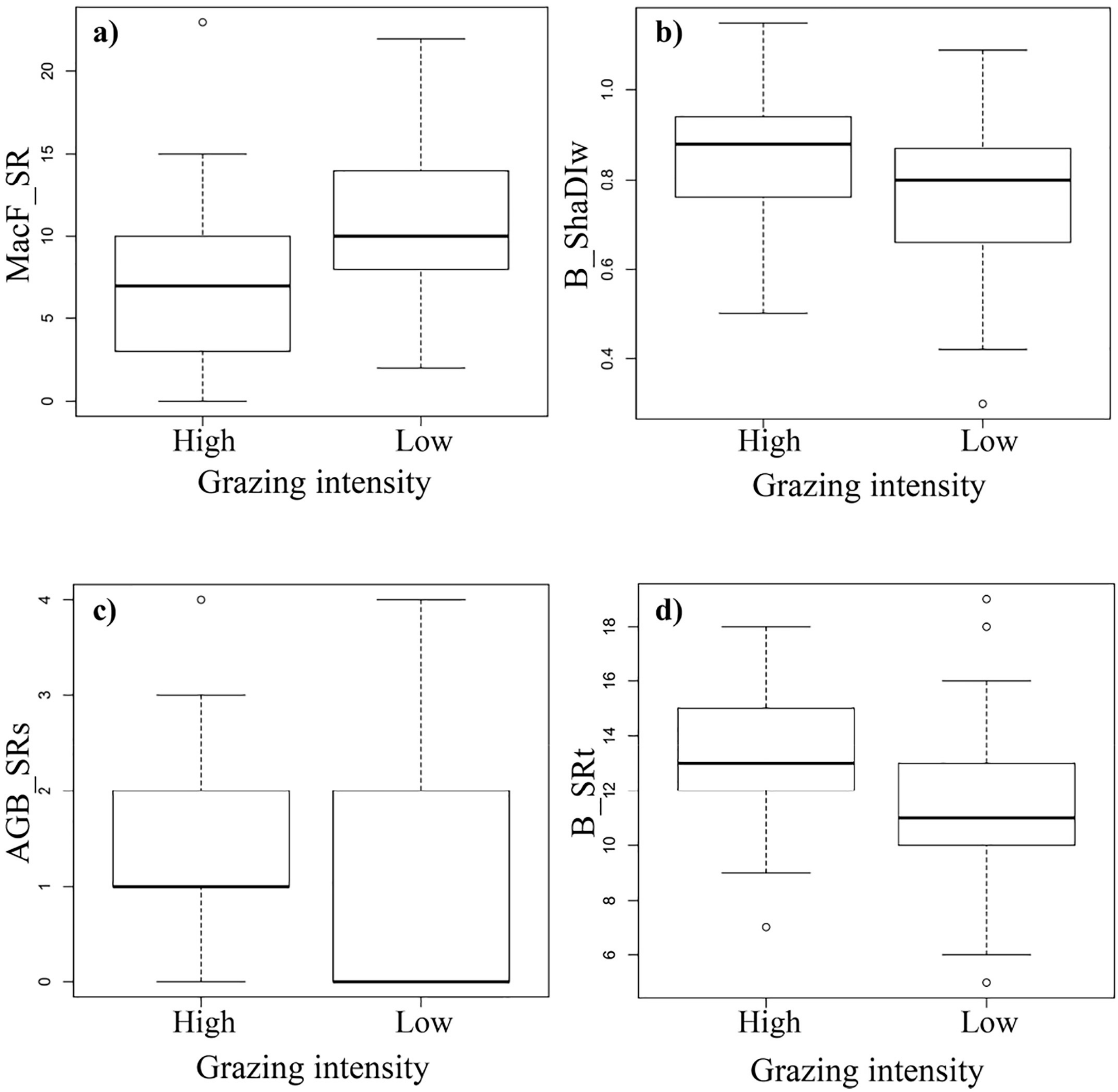

Figure 2. Box plots with the values of species richness on high- and low-intensity grazing plots for macrofungi (**a**), agricultural generalist birds in spring (**b**), birds in the two surveys (**c**), and the Shannon diversity index of birds in winter (**d**). MacF_SR indicates species richness of macrofungi; B_ShaDIw, Shannon's diversity index of birds in winter; AGB_SR, species richness of agricultural generalist birds in spring; B_SRt, total species richness of birds.

the percentage of bare soil, and with values for the enhanced vegetation index >0.25. On the other hand, the PC4 scores show a slight decline with tree density and with non − tree cover (however, in this latter case, the decrease observed is related to the few sample points) (Fig. 4d).

Assessment of Factors Limiting Cork Oak Tree Regeneration

One of the most relevant indicators of montado health is tree natural regeneration. The relative importance of different factors for cork oak tree regeneration (Fig. 5) shows that the highest value resulting from the BRT models (100 runs) is for shrub cover (~16%). Between 8% and 10% it is possible to identify different predictors: grazing intensity, mean height of the shrub layer, total cover index, and the three parameters used to assess forage quality (biomass availability, crude protein, and fiber). These forage indicators are used in this analysis as proxies for livestock feeding preferences, as explained in the methods section.

The results of the BRT models after variable reduction are presented in Figure 6 (mean total deviance: 1.923; mean residual deviance: 0.298; estimated cross-validation deviance: 0.978/standard error: 0.267; training data correlation: 0.925; cross-validation correlation = 0.633/standard error: 0.07). Tree regeneration is related to shrub cover and increases with increased values for mean shrub height and tree cover index. In contrast, higher values for tree regeneration are also related to decreasing values for forage crude protein and fiber. Tree regeneration seems to be higher in livestock-excluded patches within the

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Table 3
Results of principal component analysis with varimax rotation (with eigenvalues > 1) for the biodiversity metrics (RMSR = 0.10; fit based upon off diagonal values = 0.76; the bold values indicate absolute component loadings higher than 0.5).

Variables	PC1	PC2	PC3	PC4	*Uniqueness*
MacF_SR	**0.88**	−0.12	0.09	−0.18	*0.16*
Sapro_SR	**0.77**	0.41	−0.18	0.05	*0.21*
Rep_Pvi	−0.13	0.01	−0.02	**0.93**	*0.12*
Rep_Psa	0.25	−0.10	−0.35	0.23	*0.76*
B_SRt	0.20	**0.82**	0.10	0.32	*0.18*
AGB_SRs	−0.11	**0.86**	0.02	−0.29	*0.16*
VEG_ShaDI	−0.11	0.09	**0.85**	0.00	*0.25*
VEG_SimEI	0.46	−0.11	**0.71**	0.08	*0.26*
Eigenvalue	*1.73*	*1.62*	*1.41*	*1.14*	
Cumulative variance	*0.22*	*0.42*	*0.59*	*0.74*	

MacF_SR indicates species richness of macrofungi; Sapro_SR, species richness of saprotrophic fungi; Rep_Pvi, number of observations of *Podarcis virescens;* Rep_Psa, number of observations of *Psammodromus algirus;* B_SRm, total species richness of birds; AGB_SRs, species richness of agricultural generalist birds in spring; VEG_ShaDI, Shannon's diversity index of vegetation communities; VEG_SimEI, Simpson's evenness index of vegetation communities.

paddock or in patches where livestock intensity is lower, coinciding with lower quality (assessed as crude protein concentration) pasture and well-developed shrub and tree layers.

Discussion

Being able to assess the impact of montado grazing management on biodiversity and tree regeneration, and thus on structural diversity, is central to the application of the HNV classification in practice. High levels of biodiversity and structural diversity are central characteristics of HNV agricultural systems. In countries in the European Union, management data exist at the farm and even at the plot level in public records related to incentive programs and are updated every agricultural year. In consequence, being able to relate management models to these two types of impacts is expected to contribute valuable knowledge and a validated methodology to enable identification in practice of which montados should or should not be classified as HNV.

Factors Behind Biodiversity Distribution in Montado

We found low correlation among the species groups considered in the analysis. In accordance with previous research outcomes in other montado studies (Bugalho et al., 2011) and with similar analyses in other European silvopastoral systems (Hartel et al., 2013; Jakobsson and Lindborg, 2017; Wood et al., 2017), some groups reacted positively to high grazing intensity, others negatively.

More specifically, livestock grazing seems to have a negative impact on macrofungi, the group where impacts are most evident. Considering that average tree density was similar in all plots, variations in macrofungi can mostly be explained by the negative effects of trampling, more commonly from cattle than sheep. Trampling effects can be expected to be greater in plots grazed by cattle, which also have higher stocking densities. In contrast, birds appear to respond positively to grazing. It could be argued that this is due to the higher percentage in the samples of generalist species (both agricultural and forest) that are less vulnerable to disturbances *(lato sensu)*. Moreover, this positive effect could be related to the availability of food for seed-eating birds in the open fields used by cattle (Moreira et al., 2005), given the relation between bird diversity and areas with less tree and shrub cover revealed by the BRT models.

The influence of grazing on the diversity of natural vegetation appears less strong. This could be because the vegetation diversity in the sample plots was strongly dominated by generalist species that are tolerant of disturbance and highly adapted to a wide range of environmental conditions.

This may mean that in montados of similar composition and biophysical characteristics, at the paddock level or even the ranch level, it is difficult to identify biodiversity hot spots, where all species groups are favored. It was not possible to find a clear relation between overall biodiversity and grazing intensity at the paddock or sampling area level. Rather, considering the clear differences in the responses of different groups, biodiversity values seem to be more dependent on montado heterogeneity at the landscape and stand levels (diversity in composition, configuration, cover index, shrub cover, and the presence of larger trees), which is in accordance with the existing literature (Moreira et al., 2005; van Zanten et al., 2014; Jakobsson and Lindborg, 2017).

This is also in line with the discussion concerning the relevance of landscape heterogeneity to maintaining biodiversity in human-dominated landscapes (Benton et al., 2003; Fahrig et al., 2011). Nevertheless, the distinct influence of observations of *Podarcis virescens* in the PCA also suggests that localized events within each paddock, or within each sample area, such as the presence of dead trees or other isolated and outstanding trees, can also have a relevant role in biodiversity indexes (Manning et al., 2006; Bunnell and Houde, 2010; Jakobsson and Lindborg, 2017). Nonetheless, this relationship is poorly understood in Mediterranean silvopastoral land use systems.

Role of Livestock Grazing on Tree Regeneration in Montados

The current decay of montados, as well as the drivers of this decay, are widely covered in the scientific literature (e.g., Costa et al., 2011; Godinho et al., 2016b; Acácio et al., 2017), particularly in relation to grazing intensity (Almeida et al., 2016; Godinho et al., 2016b; Arosa et al., 2017). Long-term decline is often related to recruitment failure and tree ageing in montados and similar silvopastoral systems in Spain known as *dehesas* (Pulido et al., 2001; Plieninger et al., 2010), as well as in other European wooded rangelands (Bauer and Bergmeier, 2011). As a result, these systems have undergone a marked increase in vulnerability to new disturbances, such as fire (e.g., Acácio et al., 2009; Guiomar et al., 2015). Tree regeneration is critical, as it contributes to heterogeneous stands and heterogeneity at the landscape level, and is a determinant of the long-term sustainability of the system (Hartel et al., 2013). However, most studies so far have focused on mean livestock density at the paddock level as a proxy for grazing intensity. The main mechanisms underlying the effects of grazing in cork oak regeneration remain poorly understood, since livestock distribution within the paddocks is uneven (e.g., Barnes et al., 2008), especially in large and heterogeneous paddocks (e.g., Bailey et al., 2015).

Our results show that tree regeneration is higher in livestock-excluded patches within the paddock or in patches where grazing intensity is lower. In accordance with other studies (López-Sánchez et al., 2016; Simões et al., 2016), this is an expected result, as higher grazing intensity implies higher disturbance and lower probabilities of seedling and sapling survival.

As mentioned earlier, most paddocks in this study were grazed by cattle. Cattle prefer grasses to forbs or shrubs (Ruyle and Bowns, 1985; McKell, 1989), avoiding areas of difficult access due to either terrain ruggedness or vegetation density (Owens et al., 1991; Ganskopp et al., 2000). They prefer areas of higher-quality forage and high biomass availability (Ganskopp and Bohnert, 2009; Kaufmann et al., 2013a, 2013b). In fact, our results also indicate that higher levels of tree regeneration coincide with lower pasture quality—places where cattle can be expected to be less often compared with areas with better pasture.

Given the selectivity of livestock for areas of high forage quality, increasing the quality of pastures in areas with lower tree density may allow regeneration in areas less preferred by livestock. As von Müller et al. (2017) concluded, manipulating pastures is the

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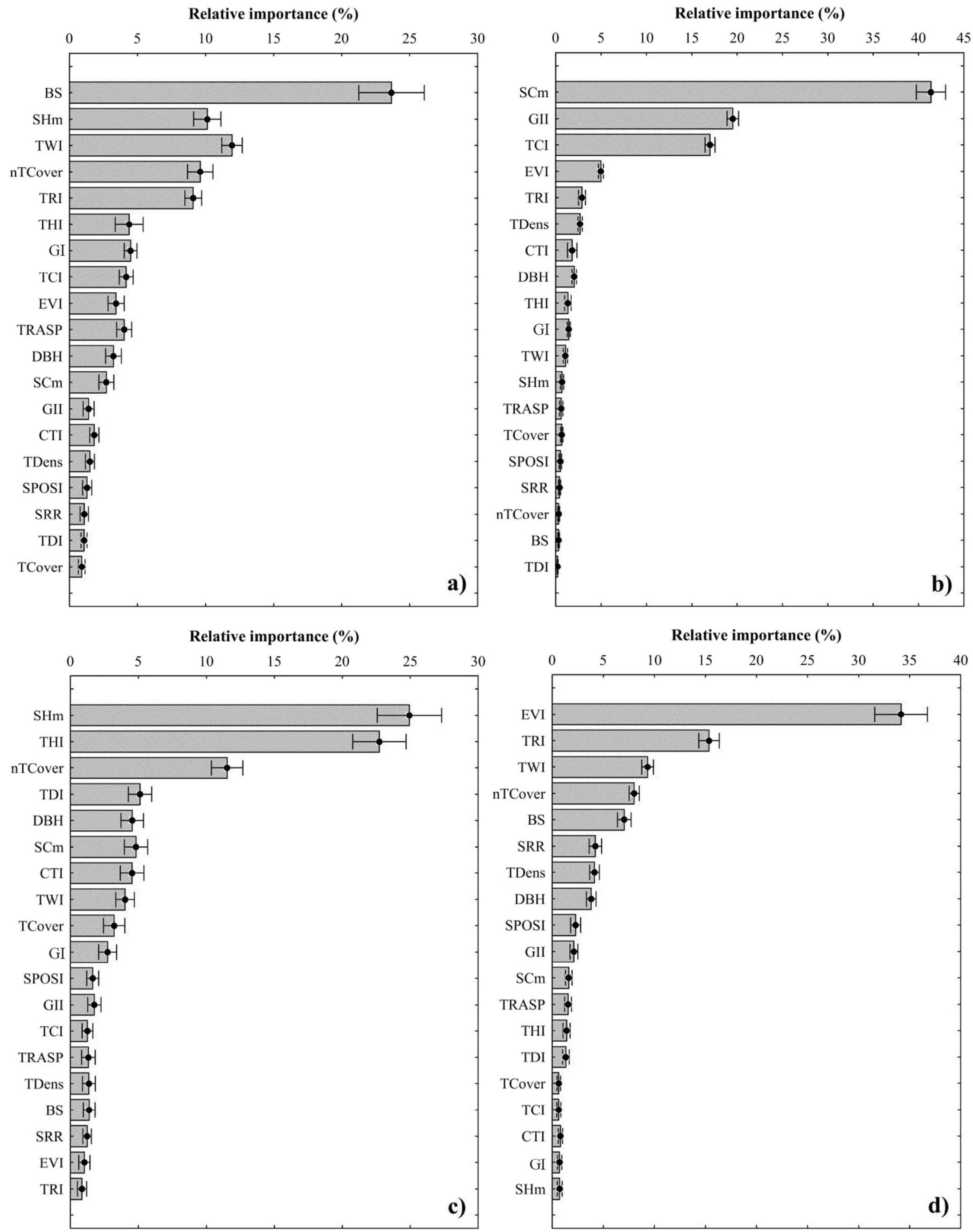
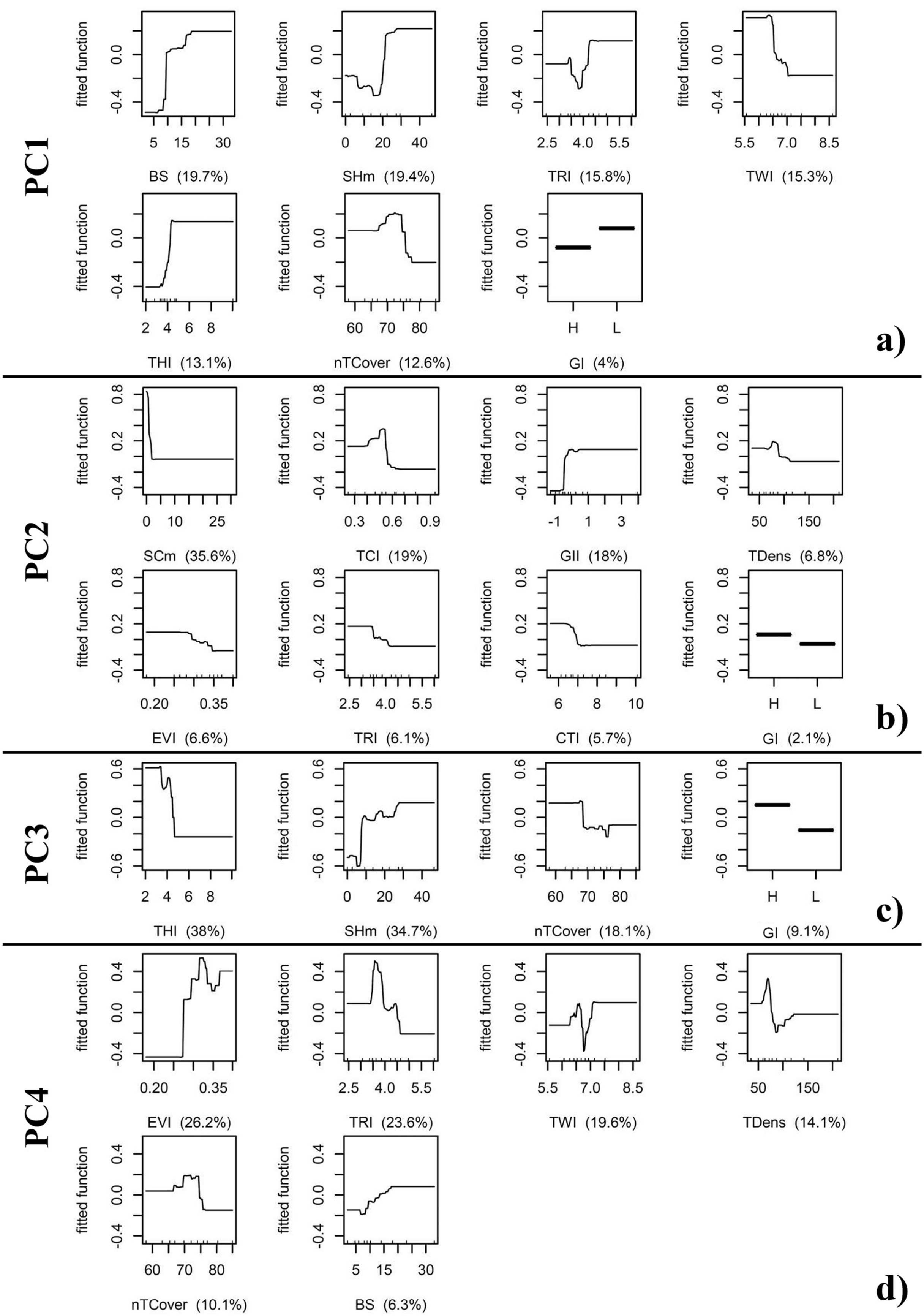


Figure 3. Mean values (±0.95 · standard deviation) of the relative importance of the predictors obtained in BRT modeling for explaining the variability in biodiversity in montado (using the scores of the 4 PCs described in Table 3 as dependent variables; independent variables were ranked by the maximum value obtained after 100 runs): **a,** macrofungi; **b,** birds; **c,** vegetation; **d,** reptiles. BS indicates percentage of bare soil; CTI, compound topographic index; DBH, diameter at breast height; DD, dung density; EVI, enhanced vegetation index; GI, grazing intensity; GII, grazing intensity index; LD, livestock density (at paddock level); NDVI, normalized difference vegetation index; nTCover, percentage of shrubland and grassland cover; SCm, mean shrub cover; SHAN, Shannon diversity index; SHm, mean height of the shrub layer; SLOPE, slope angle; SPOSI, slope position; SRR, surface relief ratio; TCI, tree cover index; TCover, percentage of tree cover; TDens, tree density; TDI, tree dominance index; THI, tree height index; TRASP, topographic solar-radiation aspect index; TRI, terrain ruggedness index; TWI, topographic wetness index.

most effective method for changing the distribution pattern of cattle grazing. In general terms, if forage availability is higher, the animals will prefer feeding on this resource and will probably be less likely to feed on the young trees. Grazing, then, is a process with diverse within-paddock influences that can contribute to landscape and paddock heterogeneity.



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Conclusion

The pertinence of the questions guiding this analysis has been confirmed: 1) biodiversity responds to management practices, particularly grazing intensity, in the montado, and thus different management practices have different impacts on biodiversity values; and 2) tree regeneration is negatively related to grazing intensity. As such, in the same type of montado, there may be areas with higher biodiversity and higher structural diversity and longevity due to greater tree recruitment. This corresponds to HNV farming characteristics, but other places in a montado might not have much in the way of HNV qualities.

Yet more nuanced conclusions can be drawn. In relation to biodiversity, it has been shown that the impact of management models cannot be usefully assessed for overall biodiversity, but only for species groups independently. For some groups, diversity decreases with grazing pressure; in others, it increases. This makes a clear-cut linkage of grazing management models to higher or lower HNV difficult to assess. Overall biodiversity values seem to be more dependent on montado heterogeneity at landscape and stand levels than at the paddock level. Our findings support the idea that maximum diversity will result from the optimum combination of plots under different levels of grazing intensity, rather than from an optimum single level of grazing intensity across all plots. In consequence, for a more accurate assessment of the HNV of montado, analysis at the plot level should be further developed and deepened by using a multiscaled approach.

For tree sustainability, the results show clearly that high grazing intensity constrains tree renewal, but within the paddock there may be differences: Depending on the quality of the pastures, within each paddock livestock will most frequently concentrate on areas with high forage quality, reducing regeneration there but allowing it in other areas. It is clear that high grazing pressure can have a negative impact on the HNV of the montado, particularly on its stability in the long run. It is also plausible to conclude that improving pastures can play a role in reducing grazing impacts on natural regeneration. Careful livestock and pasture management at the paddock level may create conditions that enhance tree regeneration in some areas while others remain more open, maintaining the HNV of the montado. Diverse tree densities will also favor higher landscape heterogeneity, which, as mentioned earlier, seems to be key in securing higher biodiversity levels and thus higher HNV. Another option for maintaining the tree stand is to plant young trees. However, this will not secure structural diversity and is not common in the montado, due to the high cost and low rate of success of new plantations, in particular in grazed areas.

Other conclusions can be derived from the analysis described in this paper. One of them is that progress was made in methodological development because a valid sampling strategy was created and used and data were jointly analyzed, both for different biodiversity groups and for their relation with real-world management options in the montado. Even though we recognize the limitations of the data obtained, the effort of combining disciplines and adapting field protocols, procedures, and boundaries contributes to progress in the construction of integrated knowledge of what is affected by different management options in the montado.

A second is in determining the limitations of the data. These limitations relate mainly to the few consistent data sets obtained, which is directly linked to the exploratory nature of this research. In future analysis, expanded field work should be foreseen, not only in the number of plots but also in the repetition of measurements over an extended time frame. Montado characteristics and conditions at a given moment are strongly determined by management activities over a long history. More long-term studies are needed. Existing public records of the management practices driven by the application of public incentive schemes were not possible to obtain for this study but would be an obvious data source for a more complete analysis.

In conclusion, we have assessed some of the relationships between management and HNV characteristics in the montado, but more study is needed if consistent identification of the high nature value of diverse montado areas is to be used for targeting public policy intervention.

Table 4
Performance of the four boosted regression tree models.

Pc	Lr	Tc	Bf	Mtd	Mrd	Ecvd (se)	Tdc	Cvc (se)
PC1	0.001	3	0.75	0.983	0.331	0.843 (0.104)	0.847	0.431 (0.115)
PC2	0.001	3	0.5	0.983	0.210	0.404 (0.081)	0.890	0.750 (0.098)
PC3	0.001	3	0.75	0.982	0.472	0.810 (0.212)	0.746	0.479 (0.072)
PC4	0.001	3	0.75	0.982	0.226	0.814 (0.072)	0.901	0.338 (0.139)

Bg indicates bag fraction; Cvc, cross-validation correlation; Ecvd, estimated cross-validation deviance; Lr, learning rate; Mrd, mean residual deviance; Mtd, mean total deviance; Pc, principal components; se, standard error; Tc, tree complexity; Tdc, training data correlation.

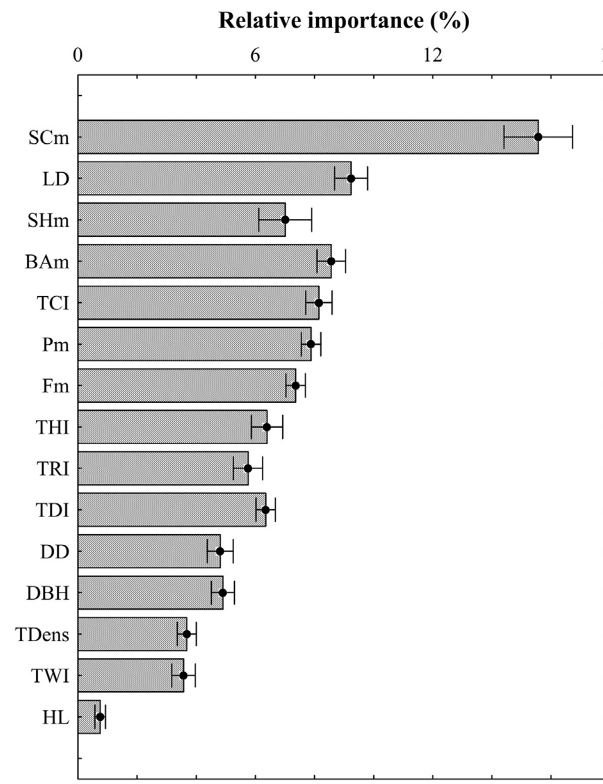

Figure 5. Mean values (±0.95 · standard deviation) of the relative importance of the predictors obtained in the boosted regression tree modeling for explaining the variability of cork oak tree regeneration (the independent variables were ranked by the maximum value obtained after 100 runs). BAm indicates mean biomass availability; DBH, diameter at breast height; DD, dung density; Fm, mean value of the forage fiber; GI, grazing intensity; LD, livestock density (at paddock scale); Pm, mean value of the forage crude protein; SCm, mean shrub cover; SHm, mean height of the shrub layer; TCI, tree cover index; TDens, tree density; TDI, tree dominance index; THI, tree height index; TRI, terrain ruggedness index; TWI, topographic wetness index.

Figure 4. Partial dependence plots for boosted regression tree models relating biodiversity principal components (already described in Table 3) to a reduced number of predictors after variable reduction through the *gbm.simplify* function. The relative importance of each variable is shown in parentheses. BS indicates percentage of bare soil; CTI, compound topographic index; EVI, enhanced vegetation index; GI, grazing intensity; GII, grazing intensity index; nTCover, percentage of shrubland and grassland cover; SCm, mean shrub cover; SHm, mean height of the shrub layer; TCI, tree cover index; TCover, percentage of tree cover; TDens, tree density; TDI, tree dominance index; THI, tree height index; TRI, terrain ruggedness index; TWI, topographic wetness index.

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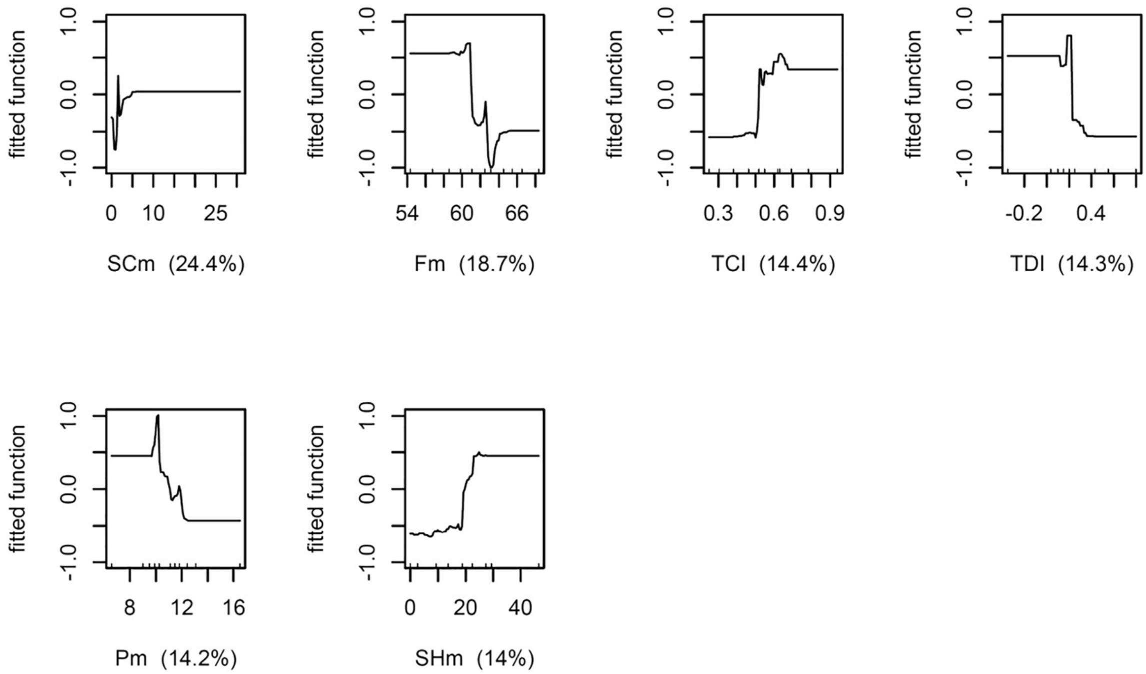

Figure 6. Partial dependence plots for boosted regression tree relating cork oak tree regeneration to a reduced number of predictors after variable reduction. The relative importance of each variable is shown in parentheses. SCm indicates mean shrub cover; Fm, mean value of the forage fiber; TCI, tree cover index; TDI, tree dominance index; Pm, mean value of the forage crude protein; SHm, mean height of the shrub layer.

Implications

The results are expected to contribute to 1) an increased awareness of montado managers of the impacts of their management on the multiple values of the montado. This awareness is particularly needed when managers seek public support in order to maintain the ecosystem services provided by montado and 2) more informed decisions on the distribution of public support, away from the simple and broad classification of the montado as HNV, and more based on relevant output criteria as proxies for the societal goals to which each montado plot contributes. The diversity of grazing effects in a wooded pasture contributes to paddock heterogeneity that supports a broad array of species. One management recommendation that we can suggest is that because livestock will most frequently concentrate on areas with high forage quality, reducing regeneration there, areas of high forage quality should be developed in more open areas to reduce intensity where regeneration is desired.

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