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Assessment of environment, land management, and spatial variables on recent changes in *montado* land cover in southern Portugal

Sérgio Godinho · Nuno Guiomar · Rui Machado · Pedro Santos · Paulo Sá-Sousa · J. P. Fernandes · Nuno Neves · Teresa Pinto-Correia

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Abstract *Montado* decline has been reported since the end of the nineteenth century in southern Portugal and increased markedly during the 1980s. Consensual reports in the literature suggest that this decline is due to a number of factors, such as environmental constraints, forest diseases, inappropriate management, and socioeconomic issues. An assessment on the pattern of *montado* distribution was conducted to reveal how the extent of land management, environmental variables, and spatial factors contributed to *montado* area loss in southern Portugal from 1990 to 2006. A total of 14 independent variables, presumably related to *montado* loss, were grouped into three sets: environmental variables, land management variables, and spatial variables. From 1990 to 2006, approximately 90,054 ha disappeared in the *montado* area, with an estimated annual regression rate of 0.14 % year^{-1}. Variation partitioning showed that the land management model accounted for the highest percentage of explained variance (51.8 %), followed by spatial factors (44.6 %) and environmental factors (35.5 %). These results indicate that most variance in the large-scale distribution of recent *montado* loss is due to land management, either alone or in combination with environmental and spatial factors. The full GAM model showed that different livestock grazing is one of the most important variables affecting *montado* loss. This suggests that optimum carrying capacity should decrease to 0.18–0.60 LU ha^{-1} for livestock grazing in *montado* under current ecological conditions in southern Portugal. This study also showed that land abandonment, wildfire, and agricultural practices

S. Godinho (✉) · N. Guiomar · R. Machado · P. Santos · P. Sá-Sousa · J. P. Fernandes · T. Pinto-Correia
ICAAM - Instituto de Ciências Agrárias e Ambientais Mediterrânicas, Universidade de Évora, Ap. 94, 7002-554 Évora, Portugal
e-mail: godinho.sergio@gmail.com

N. Guiomar
e-mail: nunogui@uevora.pt

R. Machado
e-mail: rdpm@uevora.pt

P. Santos
e-mail: aps@uevora.pt

P. Sá-Sousa
e-mail: psasousa@uevora.pt

J. P. Fernandes
e-mail: jpaf@uevora.pt

T. Pinto-Correia
e-mail: mtpc@uevora.pt

S. Godinho · N. Guiomar · R. Machado · P. Santos · J. P. Fernandes · N. Neves · T. Pinto-Correia
Departamento de Paisagem, Ambiente e Ordenamento, Universidade de Évora, Ap. 94, 7002-554 Évora, Portugal
e-mail: nneves@uevora.pt

N. Neves
e-GEO, Research Centre for Geography and Regional Planning, Faculdade de Ciências Sociais e Humanas, Universidade Nova de Lisboa, Lisbon, Portugal

(to promote pastures, crops or fallow lands) were three significant variables influencing *montado* loss.

Keywords Landscape change · Livestock grazing intensity · *Montado/dehesa* · Mediterranean · *Quercus* spp. · Spatial distribution

Introduction

The Portuguese *montado* (such as the *dehesa* land cover type in Spain) is an agro-silvo-pastoral system in which cork oak (*Quercus suber*) and/or holm oak (*Quercus* [*ilex*] *rotundifolia*) are the dominant tree species with varying densities usually in combination with livestock grazing and agriculture in the herbaceous layer (Aronson et al. 2009; Pinto-Correia et al. 2011b; Vicente and Alés 2006). *Montado/dehesa* areas account for about 3.5–4.0 Mha in the southwestern Iberian Peninsula, assuming great importance in southern Europe (Olea and San Miguel-Ayanz 2006).

The *montado* is characterised as an agroforestry multifunctional system, as it produces a range of goods and services currently in demand (Pinto-Correia et al. 2011a; Surová et al. 2011, 2014), including the following: cork; charcoal; firewood; acorns and pasture for livestock; wild game, aromatic, and medicinal plants; and recreational services, such as ecotourism (Bugalho et al. 2009; Coelho and Campos 2009; Joffre et al. 1999; Sá-Sousa 2014). According to Aronson et al. (2009), Coelho et al. (2012), Godinho et al. (2011), Plieninger (2007), and Pulido et al. (2001), the *montado* also supports other important ecosystems services, such as carbon sequestration, soil conservation, groundwater recharge and quality protection, and biodiversity conservation. Given their environmental and socioeconomic importance, *montado/dehesa* systems are regarded as high nature value farmlands (HNVF), according to European classification criteria (Almeida et al. 2013; Paracchini et al. 2008; Pinto-Correia and Godinho 2013), and are included in Annex I of the European Union Habitats Directive (92/43/CEE).

A decline in *montado/dehesa* area has been reported throughout the Mediterranean region, especially in Portugal, Spain, Morocco, France, and Italy (Brasier 1996; Cano et al. 2006; Gallego et al. 1999; Linaldeddu et al. 2013). As far as Portugal is concerned, *montado* decline has been reported since the end of the nineteenth century and is related to the intensive tree exploitation for charcoal and firewood production, man-made fires to promote open areas for crops and pastures for livestock grazing, failure in juvenile tree regeneration due to livestock browsing, and poorly understood diseases (Carvalho 1870). Surprisingly, most factors indicated in 1870 as threatening the *montado* continue to pose a threat today. The significance of such factors in the past is not known, but as they were highlighted at the time, it is assumed they must have been relevant. Paradoxically, references to threats in the past actually demonstrate the overall continuing resilience of the system, since it has survived the long term despite a number of pressures. Nevertheless, this raises the question of the limits of the *montado* in terms of sustainability: so far it has survived, but is it sustainable in the future in the face of a range of pressures of various kind (e.g. Bugalho et al. 2009; Cabral et al. 1992; Costa et al. 2010; Leitão 1902; Pinto-Correia and Mascarenhas 1999; Natividade 1950)?

Despite the observed trend of *montado* decline during the past century, the seriousness of the problem increased markedly during the 1980s (Brasier and Scott 1994; Cabral et al. 1992). A similar increase in the trend toward decline has been reported for the *dehesa* in Spain (Brasier and Scott 1994; Cano et al. 2003; Moreira et al. 2006; Plieninger 2006).

Review of the literature suggests that this decline is mainly related to the following: environmental constraints, such as soil type and hydrological conditions (Costa et al. 2008, 2010; Cubera and Moreno 2007); drought (David et al. 1992; Pelegrín et al. 2008); and wildfires (Catry et al. 2012; Díaz-Delgado et al. 2002; Silva and Catry 2006). In addition, some known diseases (e.g., *Phytophthora cinnamomi* fungus) and insect attacks also favour this decline because their effects are amplified by the already stressed conditions of the *montado* (Brasier 1996; Camilo-Alves et al. 2013; Linaldeddu et al. 2013; Moreira and Martins 2005; Pérez-Sierra et al. 2013). Furthermore, there are other factors leading to *montado* change, including the following: inappropriate management, with a sharp increase in mechanisation and unsustainable livestock stocking rates (Acácio et al. 2010; Cadima et al. 1995; Costa et al. 2010; Del Pozo 2004; Plieninger 2006, 2007); vulnerability of the agricultural economy

(Pinto-Correia 2000); rural depopulation; and the abandonment of traditional agricultural activities (Pinto-Correia 1993; Pinto-Correia and Vos 2004; Sheffer 2012). Briefly, such factors are mainly associated with changes in public policies (national and European), market mechanism and other socio-economic factors.

Several studies have examined long-term *dehesa* change in Spain (Cano et al. 2003; Plieninger 2006) and long-term *montado* change in Portugal (Acácio et al. 2009, 2010; Costa et al. 2009, 2011), but the focus was always on the local area and/or a single municipality. To support policy decisions regarding *montado* management, large-scale analyses of changes are needed to understand the overall process and to assess the role of policy.

Thus, the goal of this study is to present the changes detected in the *montado* distribution pattern for southern Portugal as a whole from 1990 to 2006 and determine the relative effects of selected environment, land management, and spatial factors on *montado* land cover change. According to the previous considerations regarding *montado* decline trends, we hypothesise that land management factors play more of a major role in causing recent *montado* change than environmental factors.

Materials and methods

Study area

This study was conducted in southern Portugal (Fig. 1), a region with vegetation dominated by cork oak (*Q. suber*) and holm oak (*Q.* [*ilex*] *rotundifolia*) species (Bugalho et al. 2009; Pinto-Correia et al. 2011b). The study area covered approximately 4.1×10^6 ha (Fig. 1), which accounts for 46 % of mainland Portugal. The selected area is in accordance with published biogeographic boundaries (Costa et al. 1998), which consist of the following several layers: phytogeographic (flora and vegetation), geomorphologic, lithologic, and pedologic, as well as bioclimate. In fact, about 72 % of the area selected is located in the so-called 'Luso-Extremadurense' province, one of the largest biogeographic provinces in the Iberian Peninsula. Its soils originate from palaeozoic siliceous material, and the original vegetation associations of Mesomediterranic cork oak (*Sanguisorbo agrimoniodis-Quercetum suberis*), holm oak (*Pyro bourgaenae-Quercetum rotundifoliae*), and pyrenean oak (*Arbuto unedonis-Quercetum pyrenaicae*) have been converted almost entirely into *montado* systems.

Cartographic data sources

The areas and distributions of *montado* land cover in 1990 and 2006 in the study area were obtained from the CORINE land cover (CLC) (scale 1:100,000) of 1990 and 2006. Due to the degree of heterogeneity of tree density and the understory uses of the *montado*, this multifunctional system does not entirely fit into a single CLC category (van Doorn and Pinto-Correia 2007). In fact, for the calculation of the CLC-1990 and CLC-2006 *montado* areas, the following land cover categories were used: '244-agroforestry areas', '311-broadleaved forest', and '313-mixed forest'. For this procedure, spatial analyses were conducted to accurately extract *montado* areas using auxiliary georeferenced data, such as the national land cover map of 1990 (LCM-1990) and the second level of the national land cover map of 2007 (LCM-2007-N2) (both at 1:25,000 scales), produced by the National Centre for Geographic Information and the Portuguese Geographic Institute, respectively. Additionally, data from the National Forest Inventories (IFN-1995 and IFN-2005) produced by the Portuguese Forestry Services as well as high-resolution true-colour orthophotomaps (2005) were also used in particular situations where uncertainty remained after the previously described processes. Thus, for 1990, all patches categorised as '*montado*' in the LCM-1990 were used, and they were intersected with the 244-, 311-, and 313-land cover categories from the CLC-1990 to extract only the proportion that corresponded to *montado* areas maintaining the cartographic characteristics of the CLC project. Finally, the same three land cover categories (244, 311, and 313, extracted from CLC-2006) were used to produce the *montado* map for 2006. These patches were overlaid with all patches classified as 'heterogeneous agricultural areas' in the LCM-2007-N2 and the *montado* areas mapped within each category. The IFN-1995 and IFN-2005 were used to reduce uncertainty in the final classification; visual interpretation and screen-digitised processing of the very high-resolution orthophotomaps served the same purpose.

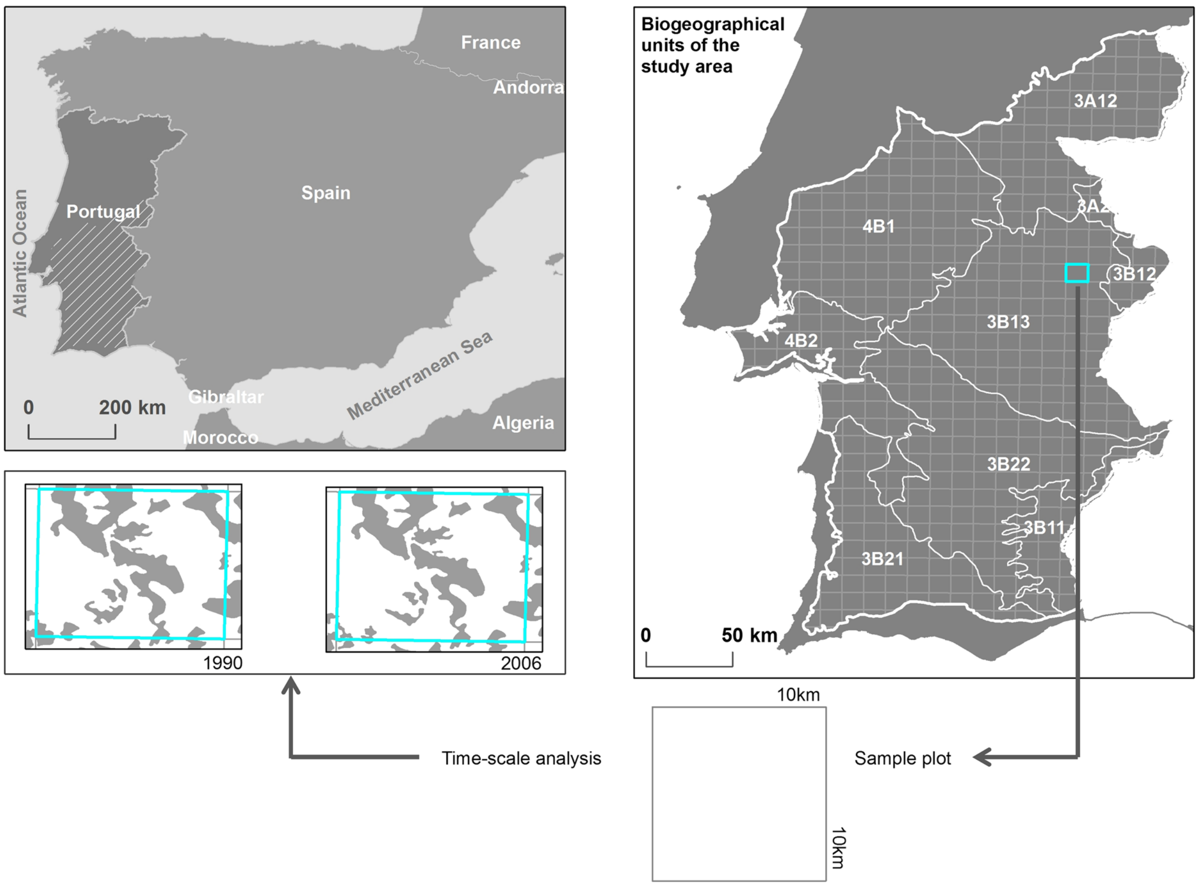

Fig. 1 Study area, biogeographic boundaries and 10 × 10 km UTM squares

Trends regarding changes in *montado* land cover during the 1990–2006 period were estimated using the UTM grid (with 10 × 10-km cell size) overlapped on each *montado* map to determine the *montado* area for each year and 10 × 10-km cell (henceforth simply referred to as cells). A total of 487 cells were examined in the study area, for which *montado* loss and gain were quantified by comparing the two maps.

Factors affecting *montado* land cover change

In the period of study (1990–2006), most of the change in *montado* land cover was attributed to loss processes, while observed gain values were insignificant throughout the study area. Thus, the analysis of recent patterns of *montado* land cover change only focusses on the loss processes.

Factors influencing *montado* loss were divided into three sets of explanatory variables: environmental (ENV), land management (LMA), and spatial (SPA) (Table 1). The stratification of these factors allows for the study of their combined effects as well as the relative influence on the spatial distribution of *montado* loss. Furthermore, the SPA set allows whether to quantify SPA influence on distribution of *montado* loss values or to correct the possible presence of spatial autocorrelation in the data (Borcard et al. 1992; Legendre 1993; Plant 2012).

Environmental set

The environmental set is composed of four variables: one of these (FIRE) represents the burnt area from 1990 to 2006, and the other three correspond to different soil fertility levels (SOIL1, SOIL2, and SOIL3) (Table 1). The GIS shapefile 'burnt areas' was provided by the Institute for Nature Conservation and Forests (ICNF). Soil fertility levels were obtained and

Table 1 Variables sets and correspondent explanatory variables description

Variable code	Description	Unit	Source
Spatial set			
AC	Autocovariate term	–	*autocov_dist* function of *Spdep* R package
X	X coordinates centered	–	GIS analysis
Y	Y coordinates centered	–	GIS analysis
XY	Multiplication of centered X and Y	–	–
X^2	Square of X coordinates centered	–	–
Y^2	Square of Y coordinates centered	–	–
Environmental set			
FIRE	Burned area due to wildfire 1990–2006	Proportion	Portuguese National Forestry Authority
SOIL1	Infertile soil	Proportion	CEEM (1996)
SOIL2	Moderately fertile soil	Proportion	
SOIL3	Fertile soil	Proportion	
Land management set			
UAA_NU	Useful agricultural area not used	Proportion	Portuguese General Census of Agriculture 1999 (GCA99)
LSTOCK	Livestock units: cattle, goats and sheep	Livestock units	
NPAST	Improved natural pasture lands under *montado* cover	Proportion	
PCFL	Pasture, crops and fallow land under *montado* cover	Proportion	
NCROP	*Montado* under cover without crops	Proportion	

adapted from the classification produced for the STRIDE-Amb. 12 final report (CEEM 1996). Fire and soils variables were superimposed on the 10 × 10-km grid layer using GIS software (ArcGis 10 and ESRI 2011), and proportional values for each variable were extracted for each 10 × 10-km cell.

Land management set

The five variables comprising the land management set were obtained from the Portuguese General Census of Agriculture (GCA) carried out in 1999 (Table 1). The GCA information is provided at the parish level. Indeed, data of the five variables were extracted for each 10 × 10-km cell by calculating the proportional area by parish within each 10 × 10-km cell. These five variables include the following: UAA_NU, the proportion of useful agricultural area not used in each 10 × 10-km cell, which was used as an indirect measure of rural abandonment; LSTOCK, the estimated grazing intensity obtained by converting cattle, goat, and sheep numbers into livestock units (LU), while the LU of each livestock type was summarised to obtain total LU per 10 × 10-km cell; NPAST, the proportion of area under *montado* cover occupied by improved natural pasture land; PCFL, the proportion of area under *montado* cover occupied by pasture, crops, and/or fallow land; and NCROP, the proportion of area under *montado* without crops.

Spatial set

The spatial set is composed of six variables: three basic (X, Y, and AC) and three derived variables (XY, X^2, and Y^2) (Table 1). It is well known that land cover data exhibit spatial autocorrelation, meaning that closest pairs of points have the tendency to be more similar than points at larger distances (Overmars et al. 2003). Therefore, regarding land cover change analysis is crucial to understand and incorporate the spatial correlations of the dependent variables in statistical models. Before statistical modelling, the existence of any autocorrelation in *montado* loss (dependent variable) was assessed using Moran's *I*. To capture the spatial autocorrelation of *montado* loss values, an autocovariate term (AC) (Dormann et al. 2007) was calculated using the R package *spdep* (Bivand et al. 2010). Moreover, for this set of variables, a second-order polynomial of centred spatial coordinates (X^2 and Y^2) was computed to capture a larger-scale spatial variation (Legendre and Legendre 1998; Miller et al. 2007).

Statistical analysis

To understand the underlying causes of recent tendencies for *montado* change, the amount of *montado* area lost in each 10 × 10-km cell from 1990 to 2006 was defined as a dependent variable.

The relationships between the dependent variable (*montado* loss) and the independent variables were analysed by means of a three-stage statistical analysis involving: (1) exploratory analysis; (2) model building; and (3) variance partitioning. The first stage was performed using exploratory plots and linear regression models for screening the response curve shape (Zuur et al. 2009). This procedure was useful for verifying if the *montado* loss values increased (or decreased) linearly with a specific independent variable. Exploratory analysis revealed that the main relationships between the dependent and independent variables were unlikely to be linear. Consequently, it was decided that the generalised additive model (GAM) (Hastie and Tibshirani 1990) should be used to assess the relationships between the covariates and *montado* loss. The GAM is more flexible than the generalised linear model (GLM), allowing for both linear and complex additive response shapes as well as a combination of the two within the same model (Wood and Augustin 2002). As with the GLM method, GAM models use a link function to establish a relationship between the mean of the response variable and a 'smoothed' function of explanatory variables.

The second statistical stage (model building) started with a univariate GAM analysis for all independent variables and predictors (Table 1). This analysis was appropriate for verifying the significance of each independent variable in explaining the *montado* loss values. Only variables with univariate significance p values <0.25 were used in posterior analyses (Tabachnick and Fidell 2001). To check multicolinearity, pairwise Pearson correlations among all predictors were computed, and pairs with $r > 0.7$ were excluded from further analyses (Tabachnick and Fidell 2001). Multivariate models were then constructed independently for each set of predictors (SPA, ENV, and LMA) using a GAM with an identity link function and Gaussian error term (Wood 2006) to select the most parsimonious models to be used in further analyses. Generalised cross validation (GCV) was used as a criterion for estimating the smoothing parameters (Wood 2006). For each set of predictors, models with all possible combinations of remaining variables (following univariate analysis) were devised and compared with Akaike information criteria corrected for small samples (AIC_c) (Burnham and Anderson 2002). Models with $\Delta AIC_c < 4$ are considered to have great relevance as candidate models (Burnham and Anderson 2002). Akaike weights (w_i) were also calculated as model selection criteria (Burnham and Anderson 2002), where the highest w_i represents the best model for ecological interpretations. The goodness-of-fit for each model was measured by means of deviance statistics (D^2) (Venables and Ripley 2002).

In the third stage, variance partitioning was used to specify which proportion of the variation in *montado* loss values is explained by each of the three factor sets exclusively as well as which proportions are attributable to interactions between factors (Borcard et al. 1992; Legendre 1993). The effects of different factors on the distribution of *montado* loss values may coincide with each other or counteract one another; therefore, the sum of the amount of explained variation by each set of variables usually differs from the total amount explained by the three sets together. Thus, seven fractions representing explained variation were obtained by means of the partitioning method: (1) the pure effect of ENV; (2) the pure effect of SPA; (3) the pure effect of LMA; (4) the shared effect of ENV + SPA; (5) the shared effect of ENV + LMA; (6) the shared effect of SPA + LMA; and (7) the shared effect of ENV + SPA + LMA. The D^2 was used as a measure of variance explained by each GAM model (Guisan and Zimmermann 2000). All statistical analyses were conducted using R 2.14.2 (R Development Core Team 2011) software, using the *mgcv* package for GAMs (Wood 2006).

Results

Factors affecting *montado* land cover change

In the study area, it was estimated that *montado* covered approximately 1,310,756 ha in 1990, while, in 2006, it decreased to 1,220,702 ha (Fig. 2a, b). In

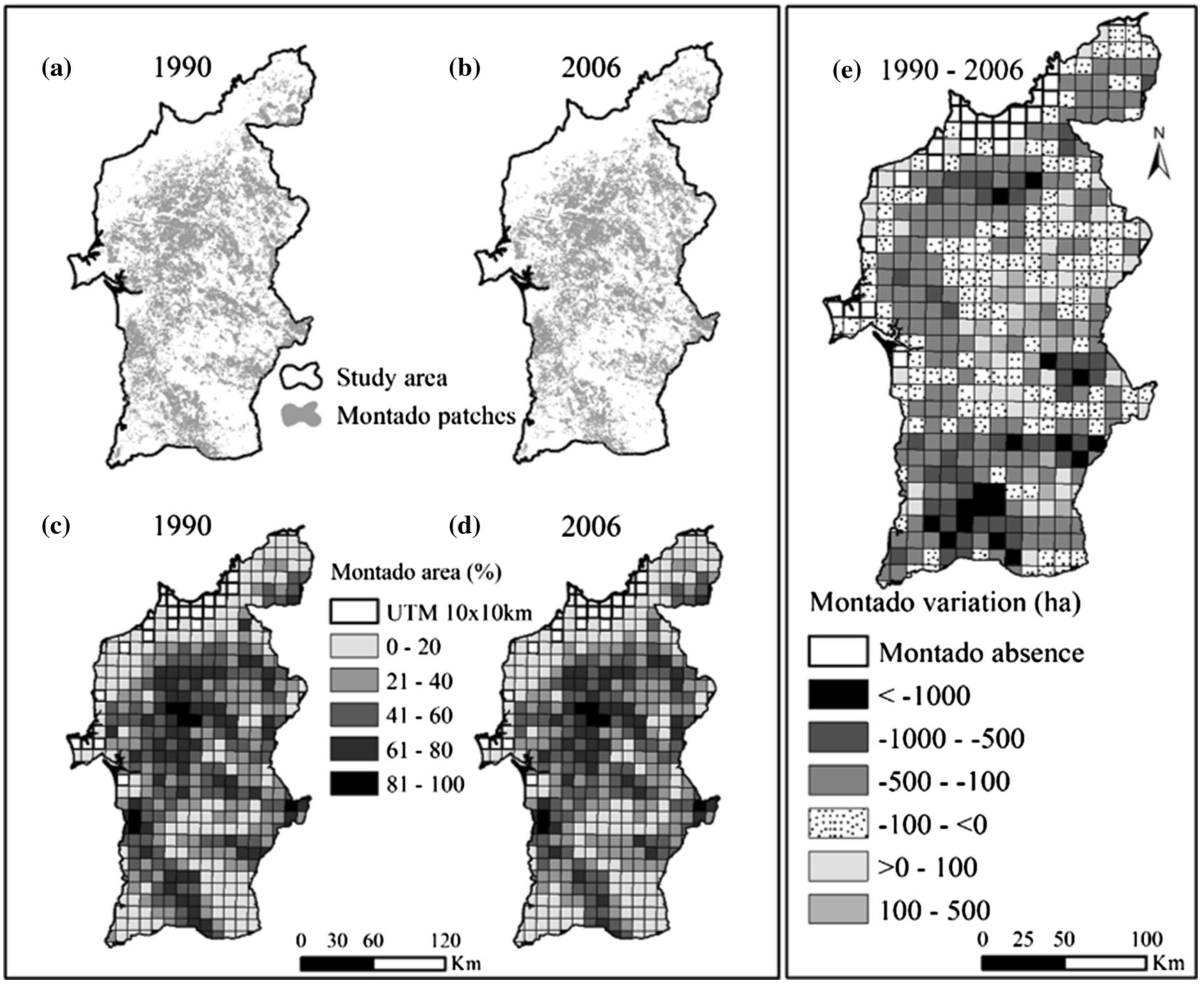

Fig. 2 Spatial-temporal patterns of *montado* landscape between 1990 and 2006. **a** *montado* area in 1990 (1,310,756 ha), **b** *montado* area in 2006 (1,220,702 ha), **c**, **d** % of *montado* area in each UTM square in 1990 and 2006, respectively, **e** *montado* area variation (in ha) between 1990 and 2006 in each 10 × 10 km cell

1990, approximately 16.3 % (from a total of 487 cells) had more than 60 % of *montado* area, while, in 2006, there was only 13.2 %, reflecting a sharp decrease during the 16-y period (Fig. 2c, d). From 1990 to 2006, no cells showed a gain in *montado* area greater than 500 ha (Fig. 2e). At the same time, 42 cells exhibited *montado* loss values ranging from 500 to 1,000 ha, while 143 cells showed a loss of 100–500 ha (Fig. 2e). During the 1990–2006 period, an area of 90,054 ha of *montado* was lost, corresponding to an annual regression rate of 5628.4 ha year^{-1}. In total, the rate of *montado* regression estimated for the period from 1990 to 2006 was, on average, 1.4 % cell^{-1} decade^{-1}.

Model selection

After univariate analysis, multicollinearity inspection, and model selection, the number of predictors in the spatial set was reduced to three (AC, X, and Y). The five land management set predictors (UAA_NU, LSTOCK, NPAST, PCFL, and NCROP) were retained. For the environmental set, the four original predictors (FIRE, SOIL1, SOIL2, and SOIL3) were also retained.

To gauge the influence of spatial variables on *montado* loss values, seven possible additive models were used with the three other variables. Of these models, only one was considered plausible ($\Delta_i < 4$), which selected with high probability the autocovariate term (AC), X, and Y

Table 2 Best candidate models for spatial, environmental and land management sets for explaining *montado* loss data

Model Set	Model	Variables contained in the model	AIC	AICc	Δ_i	AICc (w_i)
Spatial	1	AC + X + Y	−102.15	−102.11	0.00	0.96
Environmental	1	FIRE + SOIL2	−110.72	−110.63	0.00	0.24
	2	FIRE + SOIL1 + SOIL2	−110.72	−110.59	0.05	0.23
	3	FIRE + SOIL2 + SOIL3	−110.44	−110.30	0.33	0.20
	4	FIRE + SOIL1 + SOIL2 + SOIL3	−110.27	−110.06	0.58	0.18
	5	FIRE + SOIL1 + SOIL3	−109.97	−109.84	0.80	0.16
Land Management	1	UAA_NU + LSTOCK + PCFL + NCROP	−129.98	−129.81	0.00	0.45
	2	UAA_NU + LSTOCK + PCFL + NCROP + NPAST	−129.98	−129.74	0.07	0.43
	3	UAA_NU + LSTOCK + PCFL + NPAST	−126.11	−125.94	3.87	0.06

Δ_i is the AICc differences and AICc weight (w_i) is the estimated probability that a model is the best model in the set

coordinates, showing that the spatial distribution of *montado* loss values is also influenced by spatial factors (AIC$_c$ [w_i] = 0.96) (Table 2). In the model selected, the AC term plays a crucial role in the spatial distribution of *montado* loss values ($p < 0.001$) (Table 3) due to the spatial autocorrelation verified in these values (Moran's $I = 0.09$, $p < 0.001$). During the environmental modelling procedure, 15 candidate models were tested, of which only five plausible models were found to explain the variability of *montado* loss values ($\Delta_i < 4$) (Table 2). The best model shows that *montado* loss values are optimally explained by the additive effect of FIRE and SOIL2 (AIC$_c$ [w_i] = 0.24) (Table 2). Finally, in the case of land management variables, out of 31 candidate-adjusted models, only three models were selected as being plausible for explaining the variability contained in the dataset ($\Delta_i < 4$) (Table 2). Based on Akaike weights, the model with the additive effect of UAA_NU + L-STOCK + PCFL + NCROP (model 1) presented the highest value (AIC$_c$ [w_i] = 0.45) (Table 2).

The spatial model explained 44.6 % of total variation, showing a close association of the autocovariate term with the observed *montado* loss values (Tables 3, 4). The environmental model explained 35.5 % of the variation, indicating that *montado* loss values were significantly influenced by burnt area and soil quality (Tables 3, 4). The loess curve of the burnt area plot (Fig. 3a) exhibited a sharp increase in *montado* loss values, ranging from 0.40 to 0.65 of burnt area. The land management model showed the highest percentage of explained variance (51.8 %) (Table 4). *Montado* loss values were markedly influenced by LSTOCK, PCFL, UAA, and NCROP variables (Table 3). The shape of the loess curve of the LU plot shows that *montado* loss values are close to zero when the number of LU per cell ranges from 1,800 to approximately 6,000, which corresponds to a grazing intensity of 0.18–0.60 LU ha^{-1}, and rapidly increases in cells where the LU is greater than 6,000 LU cell^{-1} (Fig. 3b). Figure 3c shows that UAA_NU has a positive effect on *montado* loss values, indicating that loss values increase in cells where the percentage of UAA_NU is higher. This suggests that the abandonment of agricultural land, in particular in marginal areas, such as mountain regions, has a negative impact on the *montado* system. Finally, the loess curve of the PCFL plot shows that *montado* loss is negatively influenced by an increase in the proportion of pasture, crops, and/or fallow land under *montado* cover (Fig. 3d).

The full model accounted for 61.0 % of the explained variation (Table 4). The largest proportion of *montado* loss was accounted for by the shared effects of the land management, environmental, and spatial sets (27.3 %). The greatest pure effect was associated with the land management model (9.7 %), whereas spatial (4.7 %) and environment (3.0 %) sets had moderate influences on *montado* loss values (Fig. 4). Other shared pair effects were 1.5 and 3.7 % for ES and EL, respectively (Fig. 4). Furthermore, the shared effect of the land management and spatial set (11.1 %) indicates that the combined effects of these two sets had a considerable influence on *montado* loss values.

Discussion

As indicated previously, this study had one key objective: to analyse the comparative importance of

Table 3 Coefficients and their significance for partial and full models for *montado* loss data

Variables	Partial models			Full model		
	Linear term			Linear term		
	β	SE	p value	β	s.e.	p value
Spatial set						
Intercept	0.125	0.023	0.001***	0.231	0.025	0.000***
AC	0.635	0.064	0.001***	0.219	0.081	0.011*
	Smoother terms			Smoother terms		
	edf	F value	p value	*edf*	F value	p value
Spatial set						
s (X)	2.042	3.670	0.016*	4.147	1.830	0.104
s (Y)	4.718	2.787	0.013*	8.173	1.008	0.391
	Linear term					
	β	SE	p value			
Environmental set						
Intercept	0.294	0.011	0.000***			
	Smoother terms			Smoother terms		
	edf	F value	p value	*edf*	F value	p value
Environmental set						
s (FIRE)	5.357	14.619	0.000***	3.847	4.583	0.000***
s (SOIL2)	4.229	7.761	0.000***	$8.004e^{-09}$	0.055	0.997
	Linear term					
	β	SE	p value			
Land management set						
Intercept	0.331	0.010	0.000***			
	Smoother terms			Smoother terms		
	edf	F value	p value	*edf*	F value	p value
Land management set						
s (UAA_NU)	5.723	2.519	0.018*	5.541	2.399	0.026*
s (LSTOCK)	6.424	10.871	0.000***	7.089	2.707	0.007**
s (PCFL)	6.225	7.844	0.000***	2.079	2.102	0.120
s (NCROP)	2.956	3.429	0.069	$5.358e^{-09}$	0.133	0.999

*** $p<0.001$; ** $p<0.01$; * $p<0.05$

environment, land management, and spatial factors on recent *montado* changes.

The influence of environment, land management, and spatial factors on recent *montado* change

For the period of 1990–2006, the estimated regression rate of *montado* (0.14 % year^{-1}) obtained in this study falls within ranges previously reported by Costa et al. (2011), which were 0.16–0.22 % and 0.26 % year^{-1} for cork oak and holm oak, respectively, and those of Plieninger (2006), who reported with 0.04–0.27 % year^{-1} for holm oak. These results apparently point to an overall trend toward *montado/dehesa* decline throughout the western Iberian Peninsula, regardless of different spatial–temporal scales of analysis. The hypothesis is that decline in different parts of this area may be accounted for by the same causes.

This study demonstrates that most of the variation in recent large-scale *montado* loss is explained by land management either alone or in combination with environmental and spatial effects. Considering only pure effects, land management variables account for most of the variability in *montado* loss. An important land management and spatial component effect on *montado* loss was also observed.

Table 4 Summary of explained deviance (D^2) of all models for *montado* loss data

Models	D^2	AIC	AICc
ENV	0.355	−110.98	−110.68
SPA	0.446	−102.15	−102.11
LUM	0.518	−129.98	−129.95
ENV + SPA	0.513	−188.37	−188.30
ENV + LUM	0.563	−188.99	−188.92
SPA + LUM	0.580	−184.05	−183.98
ENV + SPA + LUM	0.610	−215.48	−215.37

Variation partitioning showed that spatial variables were also important in explaining recent *montado* loss. Spatial data, such as land cover data, have a tendency to be dependent (spatial autocorrelation), which means that when using spatial models, some variance may be explained by neighbouring values (e.g. Overmars et al. 2003; Plant 2012; Wu et al. 2009). The great spatial effect on recent *montado* loss (spatial, ES, and LS in Fig. 4) may be attributed to human disturbances not taken into account in this study [e.g., soil degradation due to the intensification of agriculture during the 'wheat campaign' that occurred during the 1930s–1960s (Baptista 1995; Stoate et al. 2001)], seed dispersal and natural regeneration, and intrinsic processes at landscape scales (relief, local water balance, etc.) (e.g. Hubbell et al. 2001; Rutherford et al. 2008). Another reason for spatially autocorrelated patterns of *montado* loss values may be spatial interactions between *montado* and other land cover/use types not examined in this study (e.g. Ramírez and Díaz 2008; Rivest et al. 2011). As indicated by the results of this

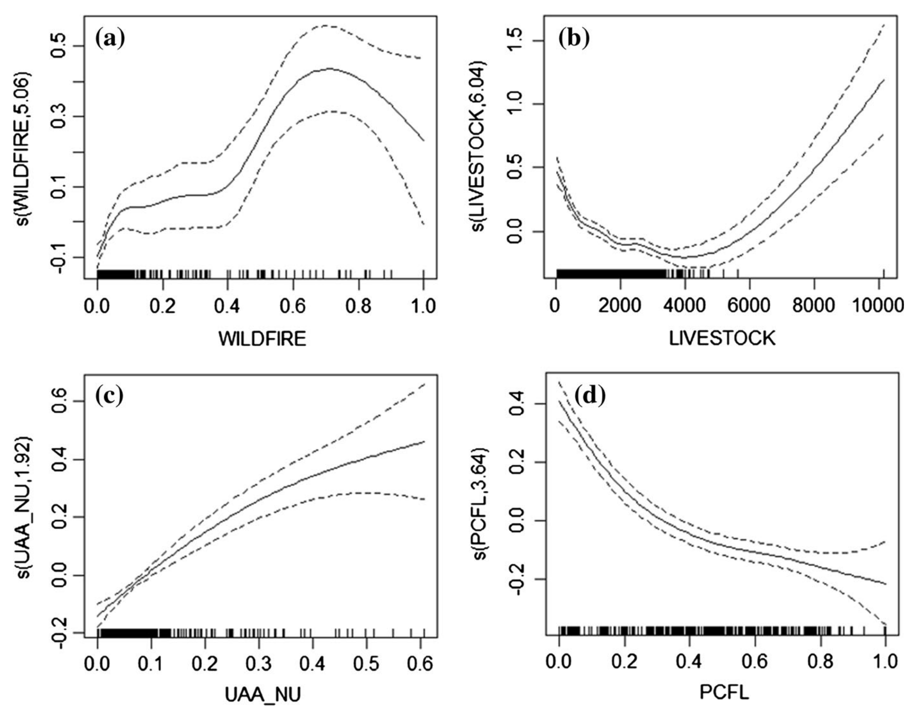

Fig. 3 Response curve shapes of **a** burned area, **b** livestock units, **c** useful agricultural areas not used, **d** Pasture, crops and fallow lands under *montado* cover in the GAM models for *montado* loss values. *X* axis represent: **a** proportion of burned area in each 10 × 10 km cell; **b** Livestock Units calculated per each 10 × 10 km cell; **c** proportion of Useful Agricultural Area not used in each 10x10 km cell; **d** proportion of area under *montado* cover occupied by pastures, crops and fallow lands in each 10 × 10 km cell. Dashed lines are approximate 95 % pointwise confidence intervals, and tick marks show the sample plots (10 × 10 km cells) along the variable range

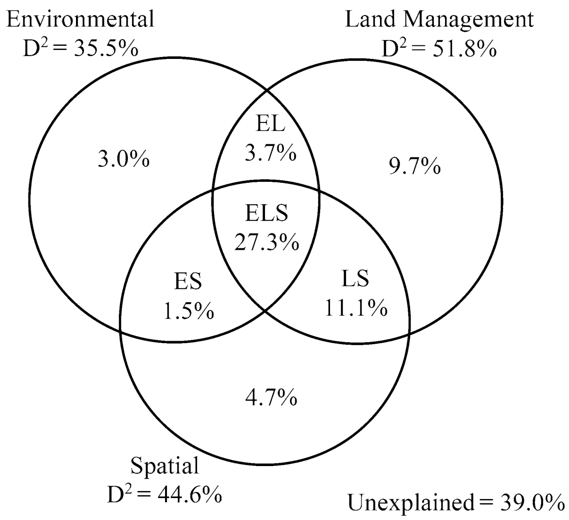

Fig. 4 Results of the variation partitioning for the *montado* loss values in terms of fractions of the variation explained. Variation in the montado loss values is explained by three sets of explanatory variables: spatial, environmental, and land use and management, and by their interactions (*EL* environment and land management (3.7 %); *ES* environment and spatial (1.5 %); *LS* land management and spatial (11.1 %); *ELS* environment, land management and spatial (27.3 %). Unexplained is the percentage of unexplained variation

study and by others on land cover change, statistical models that do not account for autocorrelation in spatial data might overestimate the importance of covariates (Lichstein et al. 2002; Plant 2012). This also might include variables that have only slight or no relevance on dependent variables (Overmars et al. 2003) and thus could lead to erroneous ecological conclusions and inappropriate management recommendations (Wu et al. 2009).

This study showed that using different intensities in livestock grazing is one of the most important variables for determining *montado* loss, as other authors have argued (e.g. Berrahmouni et al. 2007; Blondel 2006; Blondel et al. 2010; Bugalho et al. 2011; Gaspar et al. 2008; Plieninger 2007). As seen in Fig. 3b, the U-shaped curve response reflects the intrinsic relationship between the *montado* system and the different intensities of livestock grazing. This is clear evidence that ungrazed cells are associated with higher *montado* loss values, probably because these cells tend to have a more developed understory, which may have caused physiological stress due to the competition between oak trees and the understory for soil water content (Costa et al. 2010; David et al. 2007; Moreno et al. 2007). Furthermore, the water-deficit stress in *montado* trees could have been enhanced by the climatic conditions that occurred during the research period with long sequences of drier years (Costa et al. 2009; Mourato et al. 2010). Additionally, quick overgrowth of flammable shrubs (e.g., *Cistus* spp.) promotes an increased risk of severe wildfires in the absence of livestock grazing (Joffre et al. 1999). Furthermore, it was verified in the U-shaped curve that *montado* loss values were close to zero when LU per hectare ranged from 0.18 to 0.60. This suggests that the optimum *montado* carrying capacity for livestock grazing is 0.18–0.60 LU ha^{-1} in the current ecological conditions for southern Portugal. However, the selected livestock variable did not consider the duration of grazing, which is an important factor that should be considered for a more precise assessment (Calvo et al. 2012). Nevertheless, several authors showed that the carrying capacity in drier areas of the south-eastern Iberian Peninsula is close to 1.0 animals ha^{-1} or less (Baeza 2004; Calvo et al. 2012; Correal et al. 1992). Indeed, the obtained results show that *montado* loss values promptly increase when the livestock grazing intensity is greater than 0.60 LU ha^{-1}, likely indicating that a frequent overgrazing situation exists above this value. Too much grazing pressure leads to soil compaction (4.20 kg cm^{-2} in heavily grazed oak stands), which reduces water infiltration, increases water run-off, and promotes soil erosion, and leads to soil degradation (Lima et al. 2000; Pulido and Díaz 2002; Coelho et al. 2004). Furthermore, overgrazing eliminates natural regeneration of oaks due to livestock acorn predation and browsing or trampling of seedlings (Pulido and Díaz 2005).

The partial and full multivariate models used in this study also demonstrate that UAA_NU and wildfires are two important variables influencing *montado* loss. Indeed the UAA_NU is an indirect measure of rural abandonment and reflects some stress-producing socioeconomic factors that contribute to *montado* loss. In the study area, both the Algarve hills and the southern littoral Alentejo exhibited the highest UAA percentages and significant *montado* loss values. This can be explained by high depopulation rates in these areas, leading to the abandonment of agricultural land and, therefore, to gradual shrub encroachment dominated by *Cistus* spp., which can result in increased fire risk (Bernaldez 1991). These results clearly show that wildfires are a significant *montado* loss predictor, and a close positive relationship was found between burnt area percentage and *montado* loss, mainly in the Algarve hills and northern area of the study. This agrees with previous findings reported for southern Portugal, and a significant proportion of national burnt *montado* area, from 1990 to 2005, occurred in this region (Silva and Catry, 2006). Particularly, during the 2003–2005 period, a total of more than 48,000 ha of burnt *montado* area was located in the Algarve hills (Moreira et al. 2009; Silva and Catry 2006).

Finally, analysis of statistical models also showed that *montado* loss is best explained when PCFL is included in models. This variable represents land management under *montado* cover promoting pasture, crops, and/or fallow land associated with livestock production. In cells with low percentages of pasture, crops, or fallow land, high *montado* loss values were observed. This probably occurred due to the absence of land management under the *montado* and/or the combination of livestock grazing leading to greater shrub encroachment. Thus, wildfire hazard and soil water competition may cause disturbance and degradation to these ecosystems (Acácio et al. 2009; Cubera and Moreno 2007; Schaffhauser et al. 2011). Furthermore, some studies focussing on water dynamics in the *montado* system have shown that pasture and crops promoted under *montado* cover do not compete more strongly than shrubs with oak trees for available soil water resources (Cubera and Moreno 2007; Montero et al. 2004). Indeed, soil fertilisation for pastures and crops seems to favour an increase in the water-use efficiency of oak trees and an improvement of their photosynthetic rate and hydric status during the dry period (Cubera and Moreno 2007; Montero et al. 2004). To sum up, the results of this study demonstrate that the progressive disappearance of grazing at sustainable livestock levels and cereal cultivation in long rotation cycles result in shrub encroachment and subsequent *montado* decline.

Conclusion

The findings of this study strongly support the hypothesis that land management, rather than environmental factors, is the main driver of change in the condition of the *montado*, and, consequently, its spatial pattern and change. Management practices are mostly associated with the intensity and type of grazing (livestock type, breeds, density, length of time in pasture, etc.) and shrub control techniques (soil mobilisation and surface shrub cutting). On one hand, a relationship is demonstrated between loss of management and the consequent dramatic reduction in human intervention, shrub encroachment, and decay in the *montado* due to fire or the return to dense maquis-type land cover. On the other hand, it is shown how increased livestock grazing pressures also lead to *montado* decline through progressive soil compaction and prevention of natural regeneration, thus producing increasingly large areas within the *montado* that lack trees. Recent changes in management practices are closely related to the effects of the common agricultural policy (CAP), and since Portugal joined the European Union, the CAP has been the major instrument for state intervention in agricultural systems (Pinto-Correia and Godinho 2013; Primdahl and Swaffield 2010). There is, therefore, the issue of an incongruence between stated public objectives regarding the *montado* and public policies that affect this system (Pinto-Correia et al. 2014; Pinto-Correia and Primdahl 2009). On one hand, strategies related to nature conservation, cultural heritage, and tourism promote the *montado* as an important system to be preserved and enhanced due to its cultural and natural values. There is a legal protection of the trees in the system, and there are even agro-environmental schemes that foster the protection and planting of trees in the *montado* (Pinto-Correia et al. 2011b). On the other hand, coupled livestock payments are maintained in Portugal as part of the Pilar I program. This leads to the intensification of livestock production, with increased grazing intensity and a general change from sheep to cattle grazing, causing much higher impact on *montado* balance (Almeida et al. 2013; Pinto-Correia et al. 2014). During the past 15 y, these trends have resulted in strong pressures on *montado* balance, and they will be maintained most likely in the present framework program, as the same coupled payments are maintained in the present Pilar I regulation in Portugal. The findings of this study clearly show that a better balance in terms of management reduces the risk of *montado* decline. Public policy should, therefore, be revised to ensure that the range of sectoral policies remains in check with the strategies set for the preservation of the *montado* at the national level.

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