Original research article

Do the passerine traits' dynamic patterns indicate the ecological status of agro-forestry ecosystems? A modelling approach for "Montado" management assessments

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ABSTRACT

Context   The "Montado", a human shaped oak agro-forestry dominated landscape of South Western Europe has been experiencing series of changes following the implementation of different agricultural policies. Such changes are responsible for shifting the structure and composition of "Montado" landscape and hence the biodiversity of the system, either by agriculture abandonment or intensification. Traditional oak agro-forestry management practices, which focused on improving the ecological status of a single and common land use (oak agro-forestry), is taken as a conservation paradigm against the perceived changes in the system.

Aims   We aimed to identify the effect of oak agro-forestry management on "Montado" biodiversity as indicator for ecological status of "Montado" landscape.

Methods   A recently developed spatially explicit Stochastic Dynamic Methodology (StDM) was applied to model the spatial and temporal patterns of the land use/land cover changes and predict responses in biodiversity patterns, with a focus on passerine functional traits (grassland, woodland and generalist species richness), considering scenarios with and without oak agro-forestry management.

Results   Model outputs showed that oak agro-forestry management favored the expansion of oak agro-forestry at the expense of other land uses, mainly oak forest and agricultural areas. On the other hand, passerine richness exhibits a gradual decline facing the intensification of oak agro-forestry management practices, with higher declining rate observed for grassland passerine species.

Conclusion   The oak agro-forestry management does not seem to improve the ecological status of "Montado" landscape, and neither does its abandonment. Hence the conservation paradigm should focus on improving the multi-functionality of the system than merely focusing on a single and common land use

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1. Introduction

The Common Agricultural Policy (CAP) was responsible for the changing traditional agricultural patterns in Portugal (Fidalgo et al., 2007; Jones et al., 2011). This policy promotes intensive agriculture instead of traditional agriculture, which are being abandoned (Jones et al., 2011). The "Montado" ("Dehesa" in Spain) is a traditional human shaped oak agro-forestry dominated system of Iberian Peninsula that has been experiencing series of changes due to the CAP implementation, compromising land use multi-functionality and biodiversity (Beaufoy, 2013).

The "Montado" landscape comprising around 23% of forested land in Portugal (Surovà et al., 2001) characterized by a mosaic of different land use types ensured by the multi-functionality of the typical agricultural practices. This system is dominated by oaks (*Quercus suber* L. and *Quercus ilex* Lam.) and recognized as a single and generic land use type of traditional oak agro-forestry (Pinto-Correia and Godinho, 2013). The "Montado" system comprises diverse management options i.e. agriculture, oak agro-forestry including livestock farming and forestry with different ecological roles that contribute to maintain biodiversity as a whole (Beaufoy, 2013). Therefore, the idea of representing the multi-functional "Montado" system as a single land use type (oak agro-forestry) has revealed widely differing views on how and what to measure for quantifying the real complexity of such heterogeneous systems (Pinto-Correia and Godinho, 2013).

Traditional management practices in "Montado" system has contributed for maintaining the ecological status of the system. These management practices, in the context of this research, include diverse anthropogenic influences in the system, such as shrub clearance, extracting tree barks of oak for its use as wine bottle stopper, diverse agro-forestry activities in the system (Acácio, 2009). According to Pinto-Correia and Mascarenhas (1999), the main agricultural changes are either: (1) the abandonment of oak agro-forestry management or (2) the intensification of agriculture, both altering the structure and functioning of the system. Additionally, the abandonment of oak agro-forestry management is responsible for the expansion of shrub communities on non-fertile and dry lands and for the increasing area of woodland forests on fertile and humid areas (Acácio, 2009). Conversely, the agricultural intensification involving oak forest exploitation for cork, charcoal and livestock production is also responsible for compromising both its health and productivity and for reducing heterogeneity in the system (Pinto-Correia and Mascarenhas, 1999). The conservation paradigm for the entire "Montado" landscape is focused on improving the ecological status of oak agro-forestry areas through minimizing either the effect of agricultural intensification or abandonment of oak agro-forestry management (Pinto-Correia and Mascarenhas, 1999). However gearing conservation efforts only on a dominant land cover may not necessarily leads to conservation of a multifunctional landscape like "Montado". Hence, the implication of this conservation paradigm on the biodiversity and ecological status of "Montado" system is of utmost importance. This particular research is aimed at evaluating the efficiency of this conservation paradigm toward maintaining the ecological status of "Montado" landscape using robust ecological indicators.

The use of ecological indicators is crucial for investigating changes taking place in a given environment (Lindenmayer, 1999; Dale and Beyeler, 2001; Carignan and Villard, 2002; Niemi and McDonald, 2004). Although the relations between birds and agricultural changes are complex, some studies have been produced using birds as ecological indicators in agro-ecosystems (e.g. Suárez-Seoane et al., 2002; Santos and Cabral, 2004; Cabral et al., 2007; Sirami et al., 2008). In a previous overview of this problematic in Mediterranean agro-ecosystems, Santos and Cabral (2004) suggested that passerine communities present several characteristics that have justified their relevance as ecological indicators namely: (1) they usually occur in high densities in the studied habitats, (2) they are functionally placed at an intermediate position in the food webs (Moreby and Stoate, 2001), (3) they provide cheap and easy measurements (due to their conspicuous nature) if standard methodologies are applied (Bibby et al., 1992; Ralph et al., 1993), (4) they are sensitive to landscape and agricultural changes from microhabitat to landscape level (Saab, 1999; Carignan and Villard, 2002), (5) for many species, demography, behavior, distribution and phenology are connected with seasonal and spatial changes in farming practices (Ormerod and Watkinson, 2000), and (6) they have the capacity for population recovery in response to good management practices in previously disturbed ecosystems (Kati et al., 2004; Schulze et al., 2004). The use of multiple species as ecological indicators is crucial for understanding the global response of species richness to disturbance at a community level (Kati et al., 2004; Schulze et al., 2004). When these species are grouped into trophic guilds is even more efficient in reflecting the ecological response to land use gradients, both in composition and functioning terms (Santos and Cabral, 2004). The categorization of foraging behavior traits among bird species represent an added-value in order to model distinct response of birds to changes in grassland and woodland ecosystems (Preiss et al., 1997), facilitating to understand the ecological consequences, namely in terms of total bird species richness distribution, associated with the gradient of land use changes expected to occur in the study area.

With regard to how anthropogenic environmental changes will affect the abundance of species, the definition of guilds, traits or richness as indicators within communities in disturbed ecosystems is a great challenge in most ecological integrity studies (Andreasen et al., 2001). Conventionally, such potential indicators are usually estimated by using biological indices, which reduce the dimensionality of complex ecological datasets to a single univariate statistics and ordination methods (e.g. Cabral et al., 2007). Nevertheless, when the time factor is present within the data, biological indices are unable to estimate, in a comprehensible way, the structural changes when the habitat conditions are substantially changing (Jørgensen and De Bernardi, 1997). Therefore, ecological integrity studies have been improved by creating spatially explicit dynamic models that simultaneously attempt to capture the structure and composition in systems affected by long-term environmental disturbances (e.g. Bastos et al., 2012). These are, for instance, the impacts resulting from the implementation of new agricultural paradigms (Santos and Cabral, 2004).

In this scope, since many of the ecosystem phenomenological aspects are holistic, the main vocation of the Stochastic Dynamic Methodology (StDM) is a mechanistic understanding of the holistic ecological processes, based on statistical parameter estimation methods (Santos et al., 2013). This study is based on the premise that the general statistical patterns of ecological phenomena are emergent indicia of complex ecological processes that do indeed reflect the operation of universal law-like mechanisms (Cabral et al., 2007). StDM was already used, tested and validated for modeling the trend of ecological indicators in response to dynamic environmental scenarios in different contexts (Silva-Santos et al., 2006; Cabral et al., 2007; Cabecinha et al., 2009; Santos et al., 2010; Silva-Santos et al., 2010; Bastos et al., 2012; Hughes et al., 2012; Carvalho et al., 2013; Bastos et al., 2016).

The main objective of the current study is to recreate the drivers' dynamics acting on "Montado" land uses, anticipating the ecological consequences of such changes for the passerine community as a whole, expressed by the respective main functional traits under with and without oak agro-forestry management scenarios. The information obtained from this study is used to evaluate the currently operating management options and to come up with a suitable tool to support decision-making in the scope of the need for rapid, standardized and cost-saving assessment methodologies for a better conservation of the "Montado", ensuring its relevance for biodiversity.

For demonstration purposes, our approach tested two possible contrasting scenarios, which should involve either oak agro-forestry management intensification (scenario 1) or abandonment of oak agro-forestry management (scenario 2). According to CLC classification, the specific land use type that represents the typical "Montado" system is oak agro-forestry (Bossard et al., 2000; Pinto-Correia and Godinho, 2013). Therefore, for understanding the effect of management activities on the biodiversity of the "Montado", the scenario 1 (with oak agro-forestry management i.e. considering oak reforestation and shrub management) prevents any change from oak agro-forestry to other type of land uses while allowing changes from any other land use type to oak agro-forestry. Conversely, in the scenario 2 (without oak agro-forestry management) the LU/LC abandonment was simulated by changes from the typical "Montado" system (oak agro-forestry) to deep woodland (oak forest) or shrub land.

2. Material and methods

2.1. Study area

The study area is an administrative region of central Portugal, the Samora Correia county. The study area (8°56′W and 38°45′N; Fig. 1) comprises 30,578 ha of "Montado" landscape. The area is characterized by typical Mediterranean climate with "mean annual rainfall of 669 mm and mean annual temperature of 16.8 °C" (see for climate data at http://en.climate-data.org/location/507327/). The altitude of the study area ranges from 8 to 46m (see for elevation map at http://www.floodmap.net/Elevation/Elevationmap/?gi=226386). The western and northern part of the study area is dominated by areas that are under significant human influence, represented mainly by intensive agriculture (7014 ha) and built-up areas (642 ha). The southern and central part of the study area is dominated by oak forest (9887 ha), frequently interspersed by pastures, arable lands, agricultural lands, eucalyptus, and coniferous forests. The CLC information for other land covers including complex cultivation patterns, permanent crops (fruit trees and berry plantation, olive groves, vineyards), and water bodies was not extracted by the buffers around each sampling point. Therefore, these land covers (grouped here as unclassified land covers in Fig. 1) and built-up area will remain constant in our dynamic model without causing any effect on our response variables.

2.2. Bird species survey and sampling point characterization

The species data for the current study was taken from bird counts carried out on the BioAssess project. BioAssess project was a pan European project that aimed to assess biodiversity changes resulting from different land use management systems in different European countries (Silva, 2012). In Portugal, six sampling sites (Fig. 1) were selected within the study area, namely in oak forests (site 1), eucalyptus forests (site 2), in oak forests with a gradual decrease of tree density (increasing degree of fragmentation) (sites 3, 4 and 5), and in agricultural areas (site 6). Sampling within each site was based on a regular square grid of 16 sampling points (4 by 4 points spaced out by 200 m). Bird census was obtained using point counts at each sampling plot, during the years 2001 and 2002. In each year, bird sampling was undertaken in four separate visits spread throughout the passerine reproductive season (i.e. from March to June). Bird species were recorded by sound and sight at each sampling plot during five minutes (Bibby et al., 1992). Counts were made in dry weather conditions with little wind, starting at dawn or soon thereafter. During each 5 min count the observer recorded all birds, as long as they were deemed to be using the square (e.g. for nesting, displaying, hunting) into distance bands of 0–25 m, 25–50 m, 50–100 m and >100 m from the sampling point. The bird surveys were all performed by the same ornithologist.

A total of 39 passerine species identified were considered for grouping the respective richness in foraging traits (Appendix A, Table A.1). Non-passerines were excluded due to the possibility of their distribution patterns not directly connected with seasonal and spatial oak agro-forestry changes at the microhabitat/local scale. Information on the foraging trait of each species identified obtained through consulting wide range of studies including Cramp et al. (1977–1994), Sorensen (1981), Alonso et al. (1991), Tellería and Santos (1997), Favaron et al. (1998), Milwright (1998), Isenmann and
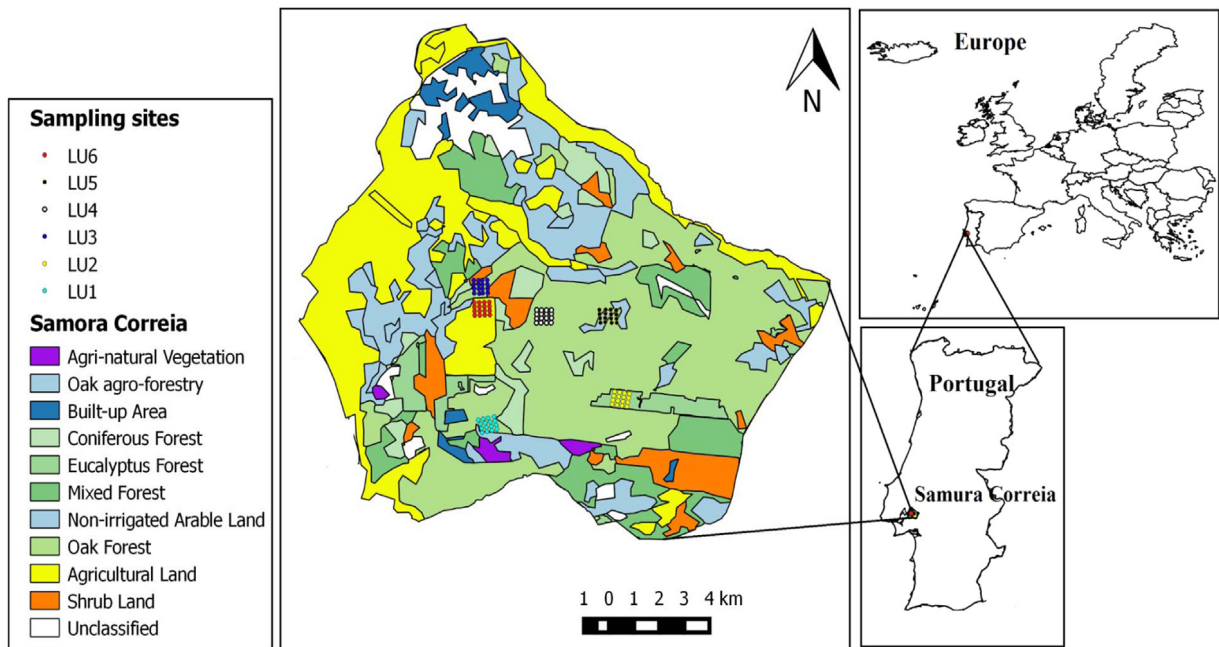
Fig. 1. (a) Location of Iberian peninsula and Portugal in Europe, (b) Location of the study area (Samora Correia county) in Portugal and (c) Location of the six selected sampling sites and land use (LU) backgrounds (LU1 — Oak forest; LU2 — Eucalyptus forest; LU 3, LU 4 and LU 5 — Oak forest with a gradual increasing degree of fragmentation and LU 6 — Agricultural areas) for field data collection on bird counts, including land use/cover variables assessment within a 500 m buffer around sampling points.

Fradet (1998), Antczak et al. (2004), Surmacki (2004) and Park et al. (2008) is used to categorize the 39 species into their respective trait groups i.e. grassland, woodland and generalist species (Appendix A, Table A.1).

A buffer of 500 m around each sampling point (Heikkinen et al., 2004) was used for LU/LC characterization using the vector format of Corine Land Cover 2000 (CLC2000, EEA, 2009), from where a total of fourteen CLC LU/LC categories were associated with our sampling points (Appendix B, Table B.1). In this context, agri-natural vegetation (AV) was characterized as areas mainly occupied by agriculture with significant extents of natural vegetation. Permanently irrigated arable lands and rice fields were considered as agricultural areas (AG), while shrub lands (including moors and heath lands) and transitional shrub lands were considered generically as shrub lands (S). Natural grasslands were considered as non-irrigated arable lands (NA). Industrial or commercial units, sport and leisure facilities were merged as built-up areas (BA). Since the CLC layer does not differentiate oak forest (F) and eucalyptus forests (E) (both described as broad-leaved forest), additional data from the COS90layer (i.e. Portuguese LU/LCmap for 1990) (DGT, 2016) was used to identify the eucalyptus forests distribution within the known distribution of broad-leaved forest in the study area. The distribution of the two oak species (*Quercus suber* L. and *Quercus ilex* Lam.) is merged as oak forest distribution since the CLC2000 does not differentiate the distribution of the two species. Additionally, a wide variety of landscape metrics (Appendix C, Table C.1) were calculated to characterize each of the 89 monitored points using Fragstats 4.2.1® (McGarigal and Marks, 1994).

2.3. Statistical analysis

Generalized Linear Models (GzLM) were used to test for the relationships between response and explanatory variables. The response variables are the richness of each passerine foraging trait considered, representative from woodland, grassland and generalist species, whereas the explanatory variables are the prevailing environmental conditions represented by the percentage of area occupied by each land use class or landscape metric (Table 1). The data summary for these metrics (Appendix D.1, supplementary materials online) is available as supplementary material online. In order to avoid multi-collinearity, predictors for the passerine traits were tested for pair-wise correlation using Spearman's rho correlation coefficient and only predictors with correlation lower than 0.7 (Wisz and Guisan, 2009) and Generalized Variance Inflation Factor lower than 5 were considered (Neter et al., 1996) (Table 1). To overcome the dependence on sample size and to allow comparability among models, the Akaike Information Criterion (AIC) was calculated for each model to rank the candidate models (Burnham and Anderson, 2002). The GzLM were run (with a Poisson variance distribution and log-link function) considering all (valid) combinations of explanatory variables for the passerine trait richness. The best model was selected according to the lower value of AIC and incorporated into the StDM model construction (Table 2). The statistical analysis was carried out using the statistical analysis software GenStat 13.0®.

Table 1
The selected land use/land cover explanatory variables (after Spearman's correlation and Variance Inflation Factor test) and their respective variable codes used in the GzLM analysis.

Selected independent variables		
Level	Variable code	Description
Patch	F_AREA	Patch area of oak forest in hectares
Class	F_PLAND	Percentage of oak forest
Class	NA_PLAND	Percentage of non-irrigated arable land
Class	AF_PLAND	Percentage of oak agro-forestry
Class	C_PLAND	Percentage of coniferous forest
Class	AG_PLAND	Percentage of agricultural areas
Class	AV_PLAND	Percentage of agri-natural vegetation
Class	S_PLAND	Percentage of shrub land
Class	F_PARA_MN	Parameter to area ratio for oak forest
Landscape	PR	Patch richness

Table 2
The regression equations, degrees of freedom (DF), the coefficient of determination (Pseudo R^2_{aj}), F-values, their significance level for all combinations reported and AIC value, as selected by GzLM analysis for foraging trait groups. The specification of all variables is expressed in Table 1.

Equation	DF	Pseudo R^2_{aj}	F	P	AIC
Richness woodland species (Rwood) $\log(Rwood) = 1.29 + 0.0109$ $(NA\_PLAND) - 0.02138$ $(AG\_PLAND) + 0.00414$ $(AF\_PLAND) - 0.0000896$ $(F\_AREA) + 0.00759 (F\_PLAND)$	5	73.55	28.14	<0.001	91.16
Richness grassland species (Rgrass) $\log(Rgrass) = 0.776 + 0.01187$ $(AG\_PLAND) + 0.0712$ $(AV\_PLAND) + 0.01074$ $(AF\_PLAND) + 0.00869$ (F_PLAND)	4	57.18	9.41	<0.001	91.5
Richness generalist species (Rgen) $\log(Rgen) = 1.120 - 0.00987$ $(AG\_PLAND) + 0.00469$ $(F\_PLAND) + 0.0107$ $(C\_PLAND) + 0.0599 (PR)$	4	61.76	10.33	<0.001	89.56

F_PLAND — Percentage of oak forest, NA_PLAND — Percentage of non-irrigated arable land, AF_PLAND — Percentage of oak agro-forestry, C_PLAND — Percentage of coniferous forest, AG_PLAND — Percentage of agricultural areas, AV_PLAND — Percentage of agri-natural vegetation, F_AREA — Patch area of oak forest in hectares, PR — Patch richness.

2.4. Conceptualization of the spatially explicit StDM

The StDM protocol proposed is based on the relationships analyzed in the previous statistical procedures, supported by datasets that include variability gradients of the most significant explanatory variables of the studied habitats. Sequences of different steps (including the preliminary land use characterization of the sampling points) are interrelated within the spatially explicit StDM framework. In a holistic perspective, the partial regression coefficients (Table 2) represent the global influence of the selected land use (explanatory) variables that are representative of several habitats and complex ecological processes in each 500 m buffer around the sampling points (Fig. 2(a)). After, three main steps are performed which include linking the outputs from statistical analysis (GzLM; Fig. 2(b)) to the dynamic model construction system (StDM; Fig. 2(c)) and the final integration of those outputs into a Geographic Information System (GIS; Fig. 2(d)) to obtain spatially explicit StDM outputs.

2.5. Dynamic model construction

In order to recreate changes in the prevalent LU/LC types for each study unit, succession rates of the representative LU/LC dynamics at the study unit level (1 km^2) were considered in the conceptual model (Fig. 3). For the dynamic model parameterization, the available data on history of land use dynamics in the "Montado" system i.e. the rates of change between the respective land uses/covers in the system determined through aerial photo interpretation of land use/cover for the last 45 years (Acácio, 2009) and 15 years (Silva et al., 2011) was used to estimate the annual rates (Fig. 3) of changes between
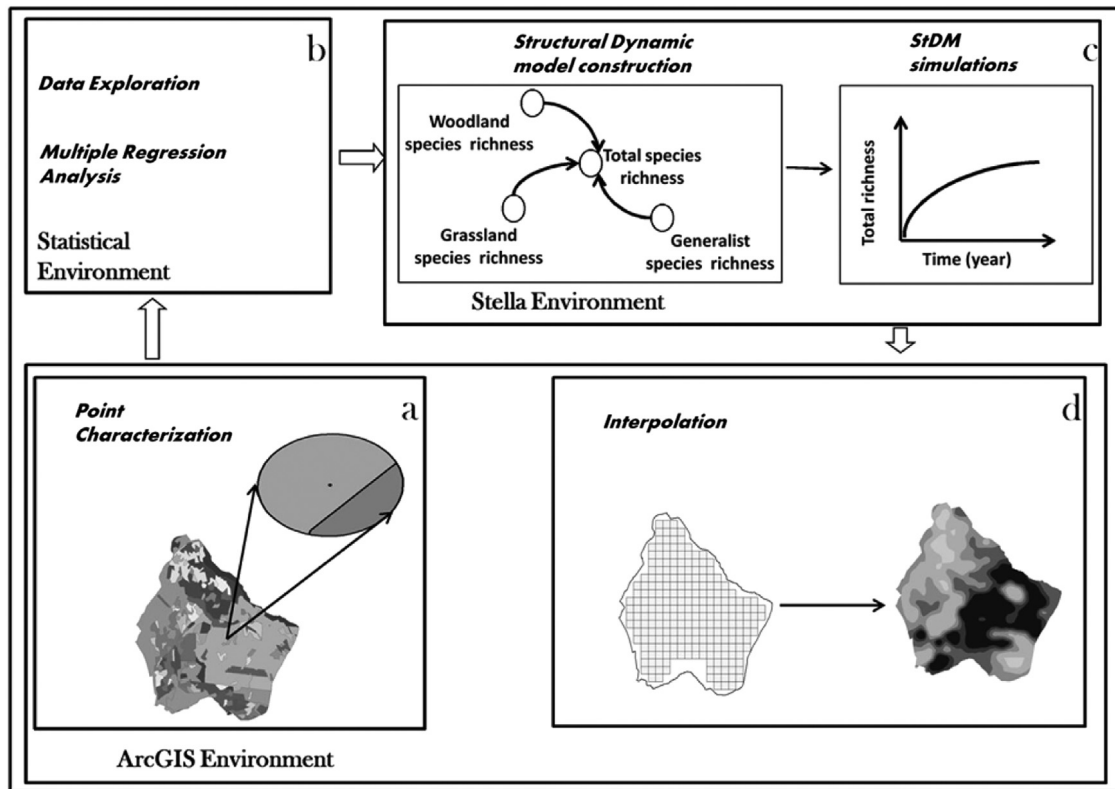
Fig. 2. The general framework of the spatially explicit StDM and the steps involved for (a) Land use characterization of sampling points, (b) Statistical multivariate regression analysis to define the convenient parameters that contextualize the habitat descriptors at lower scales, (c) Construction of the dynamic model, which allows the StDM simulations of passerine traits' responses, (d) Spatial projection of the resultant stochastic dynamic simulations per study unit and geographical interpolation enabling the creation of spatio-temporal patterns for passerine traits' responses.

the different land uses/covers. To calculate these annual rates (AR) the following formula was used (Chaves et al., 2000):

$$AR = [(1 + ATR) \exp(1/TIY)] - 1$$

where, ATR is the actual total rate inferred from Silva et al. (2011) and Acácio and Holmgren (2014) and TIY is the time interval in years (Appendix E.1, supplementary materials online). Therefore, the main transition and/or ecological succession processes among the LU/LC considered were used to mediate the condition flows between oak forest, oak savannah, and shrub community (Fig. 3).

The oak savanna (oak agro-forestry) stands represent the typical "Montado" system with understory management/grazing (Bossard et al., 2000). From the analysis of Acácio and Holmgren (2014), the oak savannas represent previous successional stages of shrub and oak forests whereas direct changes from oak forest to shrub land result from forest fire or drought in the region. The change from shrub land to oak forest occurs only if there is a strict restoration of oak forest in the region.

The basic principle of StDM protocol is given by the system dynamics balance between gains and losses of state variables, described by difference equations, which acts as environmental influences on response variables mediated by the respective partial regression coefficients. The StDM explanatory variables represent the different LU/LC types of the "Montado" landscape which are intricately linked to one another and determine the passerine species richness composition. Stochastic forest fire events were incorporated into the model to understand the effect of this potential driver on LU/LC dynamics. A fire occurrence probability once in 62 years was obtained from geographic information on annual burnt areas for the study area (ICNF, 2016). Accordingly, a random fire occurrence once in 62 years was used for the current model. The impact of fire (fire intensity) also made to vary randomly between zero (no effect) to one (total removal of a particular LU/LC within a study unit). After the calibration procedures, the average number of 4.21 ± 1.66 fires per km^2 (mean $\pm$ standard deviation) obtained per study unit in 50 years of simulation, from 100 independent stochastic simulations, was considered a reliable reproduction of the regional historical trends of fire events with an average value of 3.91 ± 2.82 fires per km^2 (mean $\pm$ standard deviation).

Different extent of fire proneness was considered since each LU/LC is characterized by specific resistance to fire (Silva et al., 2009). The fire proneness for agricultural areas and land uses/covers without forest/shrub cover i.e. agri-natural vegetation and non-irrigable arable land is not incorporated in the model since these land uses/covers are insignificantly affected by
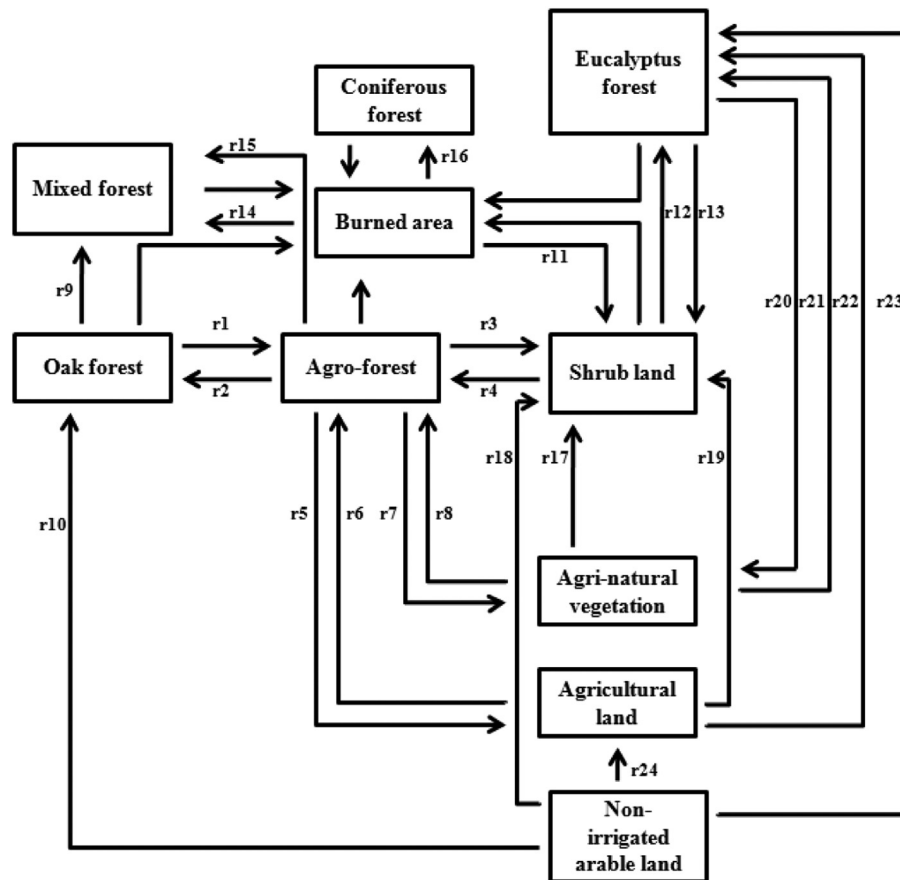
Fig. 3. The conceptual diagram showing the main land use/cover (LU/LC) dynamics between oak forest, oak agro-forestry, shrub land and other land use types with influence on the passerine traits' dynamics (StDM response variables). Arrows indicate the flows between LU/LC types, the processes (r1) to (r24) represent the estimated annual rates for the expected changes between the respective LU/LC types. Arrows without labels represent the changes from LU/LC types into burned areas which is determined by fire occurrence probability and the fire proneness of respective LU/LC types. The remaining land use i.e. built-up area is not included in the conceptual diagram assuming that it will remain the same throughout the simulation period.

forest fire (Jaiswal et al., 2002). The time unit chosen was the year and the simulation was established for a period of 30 years. StDM conceptual sub-models of land use dynamics, fire probability and burned areas, patch richness and response variables (Appendix F – Figure F.1 to Figure F.5) and the equation for explanatory (land use) variables, response variables (bird species richness) and for fire probability, fire proneness, actual total rate of change, time for change and management options used in StDM model construction (Appendix G – Table G.1 to Table G.4) are available as supplementary materials online. For the development of the StDM dynamic model, the software STELLA 9.0.3® was used.

2.6. Spatio-temporal projections

To generate stochastic dynamic projections of the passerine foraging groups distribution facing the LU/LC changes expected for the study area (comprising a total of 252 grid cells or study units), every study unit was characterized considering the available initial land cover data for the year 2000 ($t = 0$) (Appendix D.1, Supplementary materials online). Since the dynamic model simulations are influenced by stochastic factors associated with fire events, 100 independent simulations per study unit were carried out in order to capture the average local species richness trends of the passerine foraging groups, assuming the time frames of 1st, 15th and 30th year for demonstration purposes. Although the combination of all study units captures the main gradients of the habitats that characterize the study area, each spatial output per study unit only represents a preliminary discrete contribution for the global pattern of the passerine traits spatial occurrence. Therefore, in order to incorporate spatial autocorrelation among species richness per study unit (Zhang and Murayama, 2000), an ordinary kriging interpolation was selected, taking into consideration the type of data that resulted from the StDM simulations (i.e. with continuous, non-discrete distribution and the absence of normally distributed data) and the expected spatial relations between point data (Sherman, 2011) for the overall study area (regional scale). Additional study units, located at the external borders of the study area, were simulated for interpolation purposes in order to incorporate patterns

of interaction across the borders of the bounded region, thus overcoming edge effect in the study area interpolation domain (Sherman, 2011). The outputs of simulations per each grid were compiled using the center point of each grid cell or study unit. For every scenario and time selected, this procedure included the adjustment of the semi-variogram by a sensitivity analysis of the most relevant model parameters, such as, among others (i.e. nugget, partial sill, range and anisotropy). Pairs of locations were grouped into appropriate distance class to capture local variation between neighboring sample points (lag size of 0.015 km^2) (Jopp et al., 2010). Finally, we tested the fitting of the interpolated results (in relation to the modeled data) by cross validating the predictions of the spatial model with the results from the StDM simulations for each study unit (Bastos et al., 2012). This interpolation contributes to produce spatial patterns of the response variables, creating an integrative picture, in space and time, of the passerine richness' responses to the gradients of habitat changes, both with oak agro-forestry management (scenario 1) and without oak agro-forestry management (scenario 2). The spatial analyses were carried out using ArcGIS® 10.2.1 Geographic Information System software.

3. Results

3.1. Bird species richness simulation

Woodland passerine species showed an expectable positive correlation with forest related variables (F_PLAND and AF_PLNAD) and an inverse trend with agricultural areas (AG_PLAND) (Table 2). However, this foraging trait also shows positive relationships with other non-forest related land cover, non-irrigated arable land (NA_PLAND), and a negative relationship with the patch area of oak forest (F_Area). The grassland species displayed positive relationships with the influence of both forest and non-forest related variables, however agri-natural vegetation (AV_PLAND) has the predominant effect on this trait response. Generalist species showed a contrasting response to grassland species, being positively associated with forest related variables and negatively with non-forest related variables. Unlike woodland and grassland species, generalist species also showed a positive correlation with the coniferous forests (C_PLAND) and patch richness (PR) influences.

3.2. LU/LC dynamics simulation

Under scenario 1, agri-natural vegetation, oak forest, agricultural and non-irrigated arable land exhibits a declining pattern (Fig. 4). The same pattern was also evident under the scenario 2, but with higher decline rates in scenario 1. In scenario 1, oak agro-forestry showed an increase through time together with burnt areas, mixed forest and eucalyptus forest in contrast to what is observed for scenario 2. In consequence, shrub land showed a clear declining pattern through time in the scenario 1. However coniferous forest remains stable for the entire period of simulation.

For the scenario without oak agro-forestry management (scenario 2), the entire thirty year period of prediction showed a declining trend for oak forest, oak agro-forestry, agricultural and non-irrigated arable land, while shrub land, burned areas, agri-natural vegetation, mixed and eucalyptus forest showed an inversing trend. However, the coniferous forest remained stable for the whole period, similar to what was observed under the scenario 1.

3.3. Passerine species richness trends facing "Montado" contrasting scenarios

3.3.1. Trends in foraging traits

The spatial projections (Fig. 5) indicate temporal and spatial variability among different locations within the study area, for both scenarios. Species richer areas for grassland species were found in the eastern verges of the study area while for woodland species the distribution is mainly coincident with the central and western part of the study area. The projected maps for grassland species in both scenarios show a decline in richness for the most favorable areas (with more than eight species observed per grid cell). These richer areas, that were originally concentrated in eastern margin of the study area, declined leaving small areas with no more than seven species per grid cell. Under scenario 2, the same declining pattern in species richer areas was observed, although with lower rates compared with the scenario 1.

The projected maps for woodland species show a declining pattern in species richer areas for both scenarios. These areas, originally distributed in the central and western part, become more fragmented during the last period of simulation. Although both grassland and woodland species experience decline in both scenarios, the decline for grassland species is comparatively more pronounced than for woodland species.

With regard to generalist species, the projected maps showed an increasing pattern of richness distribution along time for both scenarios, although slightly more evident in scenario 1. Looking at the projected map for the three traits, the most favorable location for generalist species occurrence coincides with the interface between the most favorable areas of grassland and woodland species.

3.3.2. Trends in total species richness

The passerine total species richness was obtained by adding up the partial contributions of each trait with the intention of understanding the overall patterns of species responses, at the community level, to land use dynamics. Although the
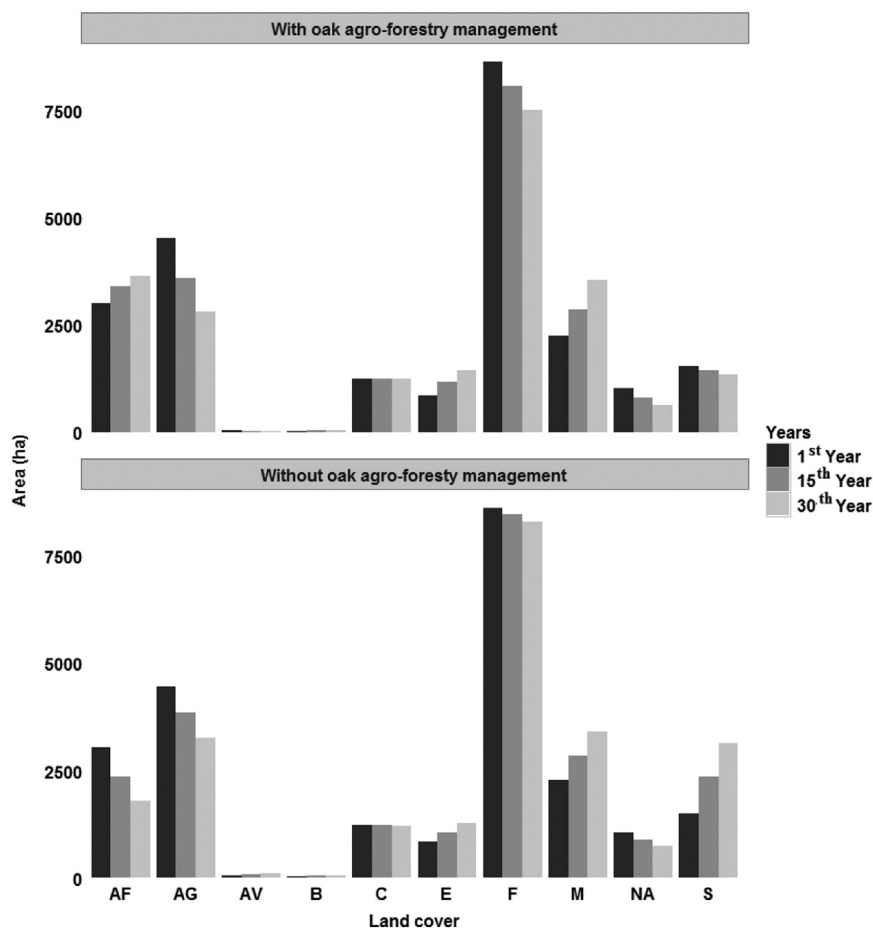
Fig. 4. The main trends of LU/LC simulated for the study area, considering the time frames of 1st, 15th and 30th year for demonstration purposes. AF — Oak agro-forestry, AG — Agriculture, AV — Agri-natural vegetation, B — Burned areas, C — Coniferous forest, E — Eucalyptus forest, F — Oak forest, M — Mixed forest, NA — Non-irrigated arable land, S — Shrub land.

average total species richness, obtained from the 252 grid cells in whole study area, showed no significant variation through time, the pattern of heterogeneity among different locations was evident in the study area, ranging from a maximum of 25 species and a minimum of 7 species. The spatial dynamic trends under scenario 1 exhibit a continuous reduction in species richness throughout the thirty years of simulation. By contrast in the scenario 2 the resultant trends show increasing number of species during the first fifteen years, followed by an inverse gradual trend until the end of the simulation.

4. Discussion

4.1. Effect of land use patterns on species diversity

In general, the land use dynamics in the study area facilitates the expansion of areas with intense and intermediate canopy while reducing the proportion of open areas. Moreover, such increase in canopy cover (oak forest, eucalyptus forest and oak agro-forestry areas) was even more pronounced when the scenario of oak agro-forestry management was implemented through avoiding land abandonment and promoting oak reforestation. Coniferous forest remains nearly constant since its dynamics is highly dependent on forest fire occurrence, which is rare in our study area (Silva et al., 2011). Other land cover types including eucalyptus forests showed increasing patterns in both scenarios due to the current increasing trend of converting unproductive agricultural areas into eucalyptus forest (Kardall et al., 1986; Jones et al., 2011).

Overall, the simulation results show that the trends simulated for the foraging traits selected were not indifferent to the habitat changes expected to occur in the studied "Montado" agro-ecosystems, where oak forest and agricultural areas represent the dominant drivers. Woodland passerines were highly associated with the area of oak forest, oak agro-forestry and agri-natural vegetation while showing negative relation with agricultural areas because the activity of most forest adapted passerines is minimal in open systems (Tellería, 2001; Suárez-Seoane et al., 2002; Sirami et al., 2008). However,
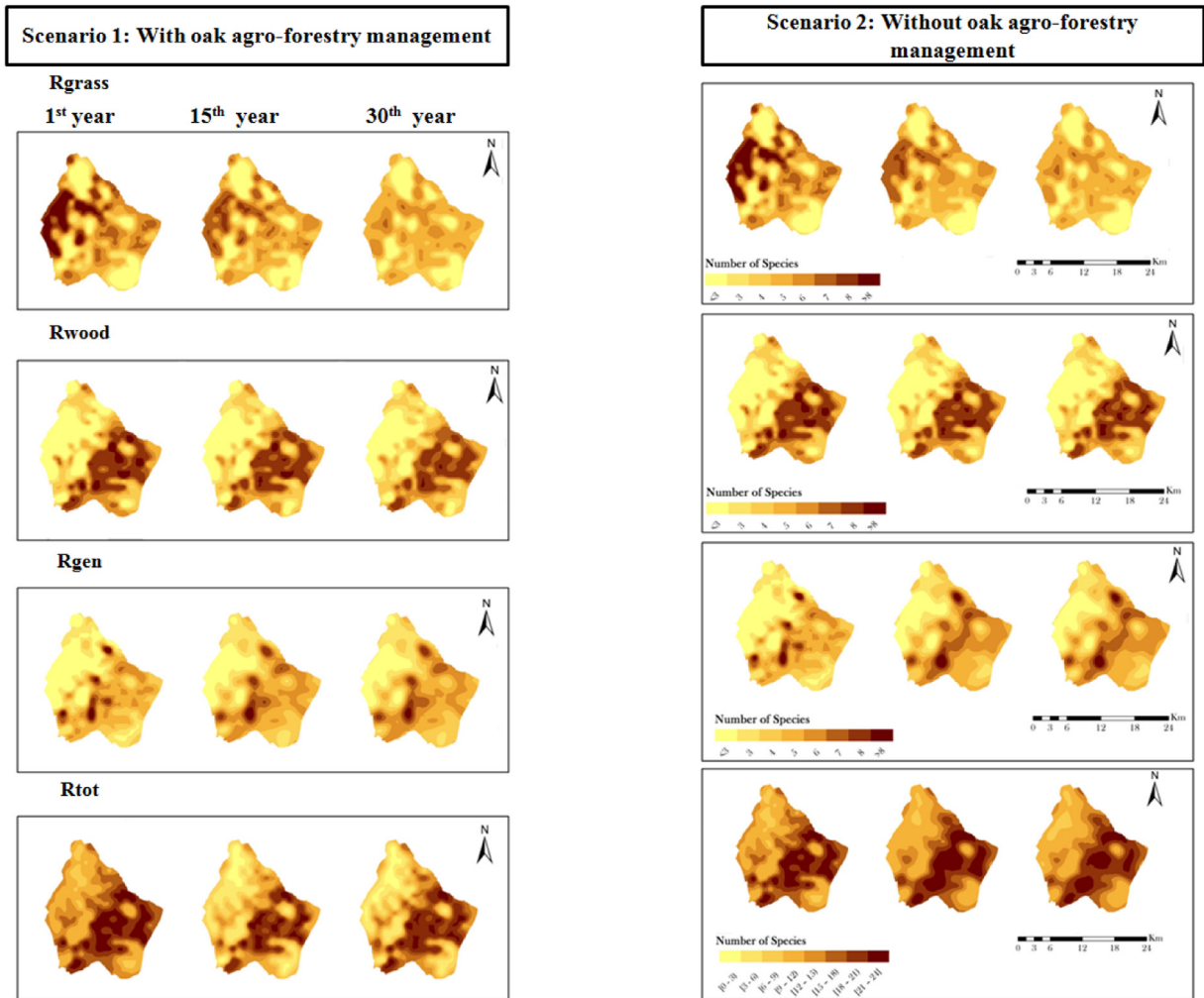
Fig. 5. Spatial–temporal patterns for the passerine foraging traits (grassland, woodland and generalist species) and total richness in the study area, considering both scenarios and the time frames selected for demonstration purposes (1st, 15th and 30th year): with oak agro-forestry management (scenario 1) and without oak agro-forestry management (scenario 2). Rgrass — Grassland species richness, Rwood — Woodland species richness, Rgen — Generalist species richness and Rtot — Total species richness.

despite the increasing trend observed for oak agro-forestry under scenario 1, a higher decline in oak forests was observed since the increasing pattern for oak agro-forestry is not just at the expense of shrub land but also some area from the oak forest is being converted to oak agro-forestry as a result of oak agro-forestry management. This main trend may explain the overall decline observed for woodland passerine species in the region (Preiss et al., 1997) also observed in scenario 1.

On the other hand, the simulations for grassland passerine species seem to be mainly influenced by changes with consequences at forest ecotones, especially between oak forests and agricultural lands, since this foraging trait also includes forest hedge species (Tellería, 2001). Other conventional studies that investigated the effects of changes in hedgerow agricultural practices on passerine bird richness have come to similar conclusions (e.g. Hinsley and Bellamy, 2000; Donald et al., 2002).

Our simulations reveal that generalist species, as well as woodland species, are positively influenced by oak forest and negatively influenced by agricultural areas. This trait is also associated with coniferous forest and total patch richness, which enables them to take advantage of the mosaic of different land cover types (Gil-Tena et al., 2007).

4.2. Spatio-temporal dynamics in the total species richness with and without oak agro-forestry management scenarios

Considering that our reference outputs at the traits level are in agreement with other studies that investigated the ecological consequences of similar agro-forestry changes, which seems to represent a credible modeling performance, the projected maps for the total species richness enable us to discuss the holistic trends at the community level for both scenarios.

The trait groups studied responded to the oak agro-forestry management scenarios simulated, which is consistent with the findings of Godinho and Rabaça (2011). However, the trait groups analyzed (grassland, woodland and generalist species) responded differently to oak agro-forestry management. In the scenario 1, the decline of grassland species was higher than other trait groups, following the pattern of open area reduction discussed previously and with clear consequences for the species richness depletion of the passerine community distribution patterns as a whole. This is because most grassland species in our region are highly adapted to Mediterranean human shaped systems like "Montado" in coexistence with managed open habitats (Tellería, 2001). Pereira et al. (2014), consistent with our findings, also indicated that oak agro-forestry management is responsible for the decline of insectivorous species, a large proportion of which area are also grassland species in our study area. However, in scenario 2 these consequences are not continuous along the thirty years period since there is a slight increase of total species richness till the fifteen year. This could indicate the importance of reducing disturbance (inherent to oak agro-forestry management intensification) to improve the biodiversity in the system. Nevertheless, further decline in richness will occur when the extent of abandonment increases in the system, namely as a result of its detrimental effect on the traditional "Montado" characteristics.

The simulation results reflect well the shift of the "Montado" oak agro-forestry toward new expected conditions and the indicators proposed are capable of responding with credibility to key changes. Therefore, considering that almost all northern Mediterranean countries are, or will be, regulated by CAP policies, encouraging the intensification of production processes, the study region will probably lose many of its traditional characteristics and the respective ecological integrity will decline (Lautenbach et al., 2011). Agricultural abandonment and intensification are just examples of the drivers and pressures affecting rural areas. In fact, either the expanding or the abandonment of oak agro-forestry at the expense of other important land covers like oak forest and agricultural areas does not result in increasing the total species richness.

An overall understanding of what is happening in "Montado" system showed the gradual decline of specialist species, which mainly comprises grassland and woodland species in Iberian Peninsula, at the expense of increasing generalist species. This may result in functional homogenization and deterioration of ecosystem structure and functioning (Clavel et al., 2011) The fact that grassland species in Mediterranean region are highly adapted to this unique ecosystem that has no its equivalent in other parts of the world will make them more vulnerable to decline and extinction (Tellería, 2001), unless suitable conservationist actions are in implementation. There is a strong need for developing indicator that could be transversally applied to all the diverse land use types in "Montado" landscape (Catarino et al., 2016). As grassland species in the context of Iberian Peninsula showed high association with the typical "Montado" landscape, they should be considered as one of the most suitable and straight forward ecological indicators for long term monitoring of environmental changes occurring in the Iberian Peninsula. Although the spatial projections recreates realistically the known passerine responses to LU/LC changes occurring in Montado regions, some limitations arise when considering validation as a fundamental process to assess the relative accuracy of the model response facing independent real data (Rykiel, 1996). This procedure was not possible in this stage of the study due to the impracticality of a credible validation for simulations that require long-term real datasets concerning future scenarios, which can only be done after several years of collecting relevant site-specific information under well-known local environmental conditions (Glenz et al., 2001; Chaloupka, 2002). However, despite these limitations, the methodology proposed seems to represent a promising starting point, allowing the development of more robust models with introduction of other indicators, interactions and interferences with precise applicable conditions to support effective management decision-making process.

4.3. Implications for land planning and biodiversity conservation strategies in oak agro-forestry systems

Land use management is often focused on few species and local processes, but in highly dynamic agricultural landscapes, only biodiversity conservation may guarantee ecological resilience (Tscharntke et al., 2005). To minimize harmful consequences of change and optimize ecosystem services, planning and management depend on the identification of relevant drivers, their corresponding scales and effects on key structures, functions and services, as well as the integration of information in multi-scale, dynamic, process-based tools for decision making (Hibbard et al., 2010).

Overall, the spatially explicit StDM framework, applied in this study, seems to represent a useful contribution to predict key changes in the foraging traits selected, namely by quantifying its main distribution area under different management scenarios. Moreover, the proposed framework allows not only the direct interpretation of the local dynamic trends at the ecological traits level, but also the visualization of their emergent spatial distribution at a community level and regional scale. The obtained simulation results seem to demonstrate the feasibility of the approach in capturing the dynamics of potential ecological indicators by predicting the behavioral pattern for key components selected under complex and variable environmental scenarios. Therefore, since habitats are characterized by a high degree of heterogeneity in space and time, influenced by many interacting factors and feedback mechanisms, this multi-scale approach is particularly helpful to capture these influences toward the definition of the "Montado" ecological status.

5. Conclusion

The management paradigm for a human shape oak agro-forestry dominated "Montado" system does not seem to improve the ecological status of "Montado" landscape. In highly diversified oak agro-forestry landscape like "Montado", the mere focus on conserving single and dominant land uses/covers does not necessarily leads to conservation of the entire

Table A.1

The list of passeriforms with their corresponding families, their common name, and foraging traits. W: Woodland, G: Grassland and I: Mixed Foragers (Foraging).

Family	Species	Common name	Foraging
Acrocephalidae	*Hippolaispolyglotta*	Melodious Warbler	I
Aegithalidae	*Aegithaloscaudatus*	Long-tailed Tit	W
Alaudidae	*Calandrellabrachydactyla*	Short-toed Lark	G
	Galeridacristata	Crested Lark	G
	Lullulaarborea	Woodlark	G
Certhiidae	*Certhiabrachydactyla*	Short-toed Treecreeper	W
Cisticolidae	*Cisticolajuncidis*	Fan-tailed Warbler	G
Corvidae	*Corvuscorone*	Carrion Crow	G
	Cyanopicacyanus	Azure-Winged Magpie	I
	Garrulusglandarius	Jay	W
Emberizidae	*Miliariacalandra*	Corn Bunting	G
Fringillidae	*Cardueliscarduelis*	Goldfinch	I
	Carduelischloris	Greenfinch	I
	Fringillacoelebs	Chaffinch	I
	Serinusserinus	Serin	G
Laniidae	*Laniusexcubitor/meridionalis*	Great Grey Shrike	G
	Lanius senator	Woodchat Shrike	G
Motacillidae	*Anthuspratensis*	Meadow Pipit	G
Muscicapidae	*Erithacusrubecula*	Robin	W
	Lusciniamegarhynchos	Nightingale	W
	Phoenicurusochruros	Black Redstart	G
	Phoenicurusphoenicurus	Redstart	W
	Saxicolatorquata	Stonechat	G
Oriolidae	*Oriolusoriolus*	Golden Oriole	W
Paridae	*Paruscaeruleus*	Eurasian Bluetit	W
	Paruscristatus	eurasianCrestedtit	W
	Parus major	Great Tit	W
Passeridae	*Passer domesticus*	House Sparrow	G
	Passer montanus	Tree Sparrow	I
	Petroniapetronia	Rock Sparrow	G
Phylloscopidae	*Phylloscopusbonelli/orientalis*	Bonelli's Warbler	W
	Phylloscopusbrehmii	Chiffchaff	W
Sittidae	*Sittaeuropaea*	Eurasian Nuthatch	W
Sturnidae	*Sturnus unicolor*	Spotless Starling	I
Sylviidae	*Sylvia atricapila*	Blackcap	W
	Sylvia melanocephala	Sardinian Warbler	I
Troglodytidae	*Troglodytes troglodytes*	Wren	W
Turdidae	*Turdusmerula*	Blackbird	I
	Turdusphilomelos	Song Thrush	I

Table B.1

The different CLC land use/land cover categories, the respective CLC codes and general description.

CLC Code	General Category	Description
121	Artificial surfaces	Industrial or commercial units
142	Artificial surfaces	Sport and leisure facilities
211	Agricultural areas	Non-irrigated arable land
212	Agricultural areas	Permanently irrigated land
213	Agricultural areas	Rice fields
243	Agricultural areas	Land principally occupied by agriculture, with significant areas of natural vegetation
244	Agricultural areas	Agro-forestry
311	Forest and semi natural areas	Broad-leaved forest
312	Forest and semi natural areas	Coniferous forest
313	Forest and semi natural areas	Mixed forest
321	Forest and semi natural areas	Natural grasslands
322	Forest and semi natural areas	Moors and heathland
324	Forest and semi natural areas	Transitional woodland-shrub
512	Water bodies	Water bodies

landscape. "Montado" system is a combination of different land uses that emanates from purely agricultural areas to mature forest. Conserving a single land cover type, oak agro-forestry, may not improve the bird species richness in a system like "Montado", where bird diversity seem to depend on a human shaped mosaic of diverse LU/LC. Rather, bird diversity could be improved through maintaining "Montado" multi-functionality by promoting the coexistence of multiple land uses in the same management paradigm. This can be achieved through understanding the patchwork relationships and recognizing the specific contribution from the combination of diverse land uses, such as those related with agriculture, forest and shrub

Table C.1
The matrices at the level of patch, class and landscape generated using Fragstats 4.2.1® software.

Landscape
Total area (TA)
Largest patch index (LPI)
Mean patch area (AREA_MN)
Mean radius of gyration (GYRATE_MN)
Mean shape index (SHAPE_MN)
Mean fractional dimension index (FRAC_MN)
Mean perimeter-area ratio (PARA_MN)
Mean related cicumscribing circle (CIRCLE_MN)
Mean contiguity index (CONTIG_MN)
Patch richness (PR)
Patch richness density (PRD)
Shannon's diversity index (SHDI)
Simpson's diversity index (SIDI)
Modified simson's diversity index (MSIDI)
Shannon eveness index (SHEI)
Simpson's eveness index (SIEI)
Modified simpson's eveness index (MSIEI)

Class
Total area (TA)
Percentage of landscapes (PLAND)
Number of patches (NP)
Patch density (PD)
Largest patch index (LPI)
Largest shape index (LSI)
Mean patch area (AREA_MN)
Mean radius of gyration (GYRATE_MN)
Mean shape index (SHAPE_MN)
Mean fractional dimension index (FRAC_MN)
Mean perimeter-area ratio (PARA_MN)
Mean related cicumscribing circle (CIRCLE_MN)
Mean contiguity index (CONTIG_MN)
Clumpiness (CLUMPY)
Interspertion juxtapositon index (IJI)
Patch cohesion index (COHESION)
Landscape division index (DIVISION)
Effective mesh size (MESH)
Splitting index (SPLIT)
Aggregation index (AI)
Normalized LSI (NLSI)

Patch
Patch area (AREA)
Patch perimeter (PERIM)
Radius of Gyration (GYRATE)
Perimeter-area ratio (PARA)
Shape index (SHAPE)
Fractal dimension index (FRAC)
Related circumscribing circle (CIRCLE)
Contiguity index (CONTIG)

lands. A clear definition of what "Montado" is still necessary to understand its complexity and for managing the balance between the diverse land uses in order to maintain the multi-functionality of the system. Therefore a new conservation paradigm considering the land use/land cover diversity and its contribution for conserving the biodiversity of "Montado" landscape is of at most importance to improve the ecological status of this threatened landscape.

Acknowledgments

Appendix A

See Table A.1.

Appendix B

See Table B.1.

Appendix C

See Table C.1.

Appendix D-G. Supplementary data

Supplementary material related to this article can be found online at http://dx.doi.org/10.1016/j.gecco.2016.09.001.

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