



Using graph theory to analyse and assess changes in Mediterranean woodland connectivity

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Abstract

Context The Portuguese *montado* is an agro-silvopastoral system, similar to the Spanish *dehesa*, known for its cultural, economic and ecological value. Despite its importance, contrasting processes such as land abandonment and land use intensification, together with several other factors, have been responsible for *montado* degradation in the last decades. Biodiversitywise, assuring high levels of connectivity is vital for many species that, in turn, contribute to the natural processes on which a healthy and sustainable *montado* relies.

Objectives To study the *montado* connectivity in the recent decades and infer what the changes represent to the short and medium dispersal species regarding habitat availability.

Methods The study was conducted in an area delimited by biogeographic boundaries in Southern Portugal where *montado* is abundant. We used a graph theory based approach and *montado* maps of 1984, 1999 and 2014 derived from remote sensing.

Results The results show a loss of *montado* associated to increasing fragmentation over time. This led to a global connectivity decrement likely to have negative implications for *montado* species. The most affected species are those more dependent on habitat characteristics, such as forest specialist birds, and those with low mobility that have lost great amounts of habitat not only due to *montado* loss but also due to the increasing fragmentation that makes suitable patches unreachable.

Conclusions Given the *montado* environmental relevance, measures should be taken in order to stop its loss and preserve the core areas that have guaranteed the connectivity over time.

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Introduction

Landscape connectivity, defined by Taylor et al. (1993) as the degree to which the landscape facilitates or impedes the movement among resource patches, is vital for ecosystems stability (With et al. 1997; Collinge 1998). Connectivity has direct influence in animal movements and population persistence (Johnson et al. 1992) and thus, is a major issue for wildlife survival and biodiversity conservation (Fahrig and Merriam 1985). Concerning plant populations and communities, different responses can be found to landscape/habitat connectivity depending, on their character (positive or negative), of the plant functional traits, life stages, matrix permeability and/or disturbance regime (Ibáñez et al. 2014). Although some species may benefit from patch isolation as a result of decreasing predation frequency or herbivory intensity (e.g. Farwig et al. 2009), the negative effects are more prevalent. Patch isolation may negatively affect genetic flow (e.g. Steffan-Dewenter and Tscharntke 1999; Seltmann et al. 2009) or seed dispersal (e.g. Kiviniemi 2008; Herrera and García 2010); and barochoric (gravity-dispersed plants) and self-incompatible species are particularly susceptible to fragmentation (McEuen and Curran 2004; Aguilar et al. 2006; Lopes and Buzato 2007). However, zoochoric plant species may also present limitations as a result of connectivity loss if they are dispersed over short distances (French et al. 2011) or if the landscape matrix is not sufficiently permeable to allow animal movements (Eycott et al. 2012; Astudillo et al. 2019).

Connectivity is central to the long-term persistence of Iberian agro-silvopastoral landscapes dominated by cork and holm oaks (e.g. Puerta-Piñero et al. 2012), in the same way these land use systems (hereafter *montados* or *dehesas*, designation used in Portugal and Spain, respectively) are critical to achieve sustainable territorial development in biophysically constrained regions due to the multiple products, values and services they provide (Martín Vicente and Fernández Alés 2006; Surová et al. 2018). The low intensity and balanced management associated with

the different vegetation layers promotes an heterogeneous landscape harboring biodiversity of high conservation value (Bugalho et al. 2011; Godinho et al. 2011), in accordance to the rule

“The low intensity greater the habitat variety, the greater the species diversity” (Rosenweig 1995). For instance, more than 95% of the terrestrial mammal species of continental Portugal can be found, occasionally or frequently, in *montados* (Pinto-Correia et al. 2013).

In this context, *montados* and *dehesas* are recognised as High Nature Value Farming systems, according to the European Environmental Agency classification (Paracchini et al. 2008; Pinto-correia et al. 2018). Despite these evidences, the sharp decline of these unique open oak woodlands persists and their long-term sustainability is threatened due to system simplification resulting both from land use intensification in the more productive areas and land abandonment of the marginal ones (Pinto-Correia and Mascarenhas 1999; Godinho et al. 2016c).

Montado fragmentation and lack of tree regeneration can result directly from the use of heavy machinery or due to grazing intensity above the carrying capacity of the available resources (Dinis et al. 2015; Almeida et al. 2016; Arosa et al. 2017; Pinto-Correia et al. 2018). The cumulative and interactive effects of ecological and anthropic disturbance regimes over a long time period have affected significantly the regenerative capacity of Mediterranean ecosystems (Blondel 2006) and, therefore, multiple recruitment limitations are also evident in less intensively used areas covered by these evergreen oak systems (Acácio et al. 2007).

Several authors argued that poor seed dispersal, low seedling survival due to predation or long-term drought, and scarcity of viable seeds are among the constraints to tree recruitment (e.g. (Leiva and Fernández-Alés 2003; Gómez-Aparicio et al. 2004; Pulido and Díaz 2005). The dominant overstorey species (*Quercus rotundifolia*—or *Q. ilex*—, and *Q. suber*) are mast-seeding evergreen trees (Pérez-Ramos et al. 2010) and their dispersion occurs through barochory and synzoochory mainly by the European jay (*Garrulus glandarius*) and the wood mice (*Apodemus sylvaticus*) (Gómez 2003; Muñoz and Bonal 2007). Puerta-Piñero et al. (2012) stress that patch connectivity has opposing indirect effects on holm oaks recruitment and survival, by increasing the

activity of acorn dispersers like jays, but also of seed predators such as wild boars, which also have negative effects on the abundance of acorn-dispersing rodents (Muñoz et al. 2009). However, low patch connectivity seems to be more critical for acorn dispersion than for predation since the negative effects of wild boars on the abundance of holm oaks seedlings and saplings can be seen throughout the landscape and thus, are less dependent of patch connectivity (Puerta-Piñero 2010).

Management type and intensity can, in turn, mediate the effect of ecological disturbances on *montado* (Acácio et al. 2017), both abiotic such as drought (Camilo-Alves et al. 2017) and fires (Guiomar et al. 2015), and biotic such as plant pathogens (Camilo-Alves et al. 2013) or insect pests (Tiberi et al. 2016). High-intensity large fires are function of the fuel connectivity both at the stand (Fernandes 2009) and landscape (Fernandes et al. 2016) levels and, therefore, *montados* and *dehesas* can act like fire-friction landscapes in the fullness of their productive functions (Azevedo et al. 2013; Fernandes 2013; Guiomar et al. 2015) since understory management allows the maintenance of low fuel loads. Shrub encroachment is more likely to occur in the more fragmented areas (Acácio et al. 2007; Guiomar et al. 2015), increasing wildfire susceptibility and also the probability of extreme fire behavior under severe weather conditions. Thus, abandoned and fragmented *montados* and *dehesas* are not only less resistant to fire spread, but also less resilient (Guiomar et al. 2015).

As other wood pastures and agroforestry systems throughout Europe, tree aging and recruitment failure must be reversed to guarantee the persistence of these multifunctional landscapes (Bergmeier et al. 2010; Plieninger et al. 2010; Roellig et al. 2018). The traditional “land sparing” strategies are insufficient to reverse the current trend of decline of these multifunctional landscapes and should be complemented by “land sharing” strategies (Grass et al. 2019), to adjust land use intensity by the carrying capacity of the system (García de Jalón et al. 2018). However, landscape-scale approaches must be promoted to define priorities for conservation and operationalise concepts such as the High Nature Value Farming systems. From the biodiversity stand point, it is also essential to monitor the *montado* spatial–temporal dynamics as it supports a large variety of species. To achieve this, the assessment of the relative contribution of the different *montado* patches to their global

connectivity, and their evolution over time, is a critical requirement.

Graph theory based approaches (Urban and Keitt 2001; Foltête et al. 2014) have been widely used in connectivity studies to approach distinct topics such as forests, protected areas, wildlife corridors design and ecological restoration (Saura et al. 2011; Pirnat and Hladnik 2016; Santini et al. 2016; Pereira et al. 2017; Hofman et al. 2018; Volk et al. 2018). Not surprisingly, a variety of connectivity related tools has also been developed to fill the researchers’ needs (Fuller and Sarkar 2006; Saura and Torné 2009; Foltête et al. 2012; Mestre et al. 2019).

In this study we use graph theory to analyse the multi-temporal *montado* connectivity in the years 1984, 1999 and 2014 with the goal to find out what the changes represent to the short and medium disperser species in terms of habitat availability. Based on the literature, which reveals a *montado* loss trend (Godinho et al. 2016c), we anticipate there has been a loss of habitat affecting many *montado* faunal species and that low mobility species may be particularly affected since their low mobility implies a lower habitat availability. In the end, we intent to provide biodiversity related information useful for future *montado* planning and management.

Methods

Study area

The study area covers 856,720 hectares in Southern Portugal (Fig. 1) (centre of the study area: 388 440 27.6000 N; 78 410 31.2000 W), in a region where *montado* is a typical land use system (Bugalho et al. 2009). The Portuguese *montado* is a multifunctional agro-silvopastoral system, similar to the Spanish *dehesa*, that covers most of the Southern region of the country, Alentejo (Pinto-Correia et al. 2011). The tree cover is composed of cork oaks (*Quercus suber*) and/or holm oaks (*Quercus rotundifolia*) in different densities (Godinho et al. 2016a) and the management practices also shape the understory that may be cultivated or a natural pasture (Canteiro et al. 2011). Thus, the *montado* is a complex system that depends not only on site variability (soil, climate, topography, etc.) but also on the conjunction of production activities (agriculture, livestock, etc.) that share the

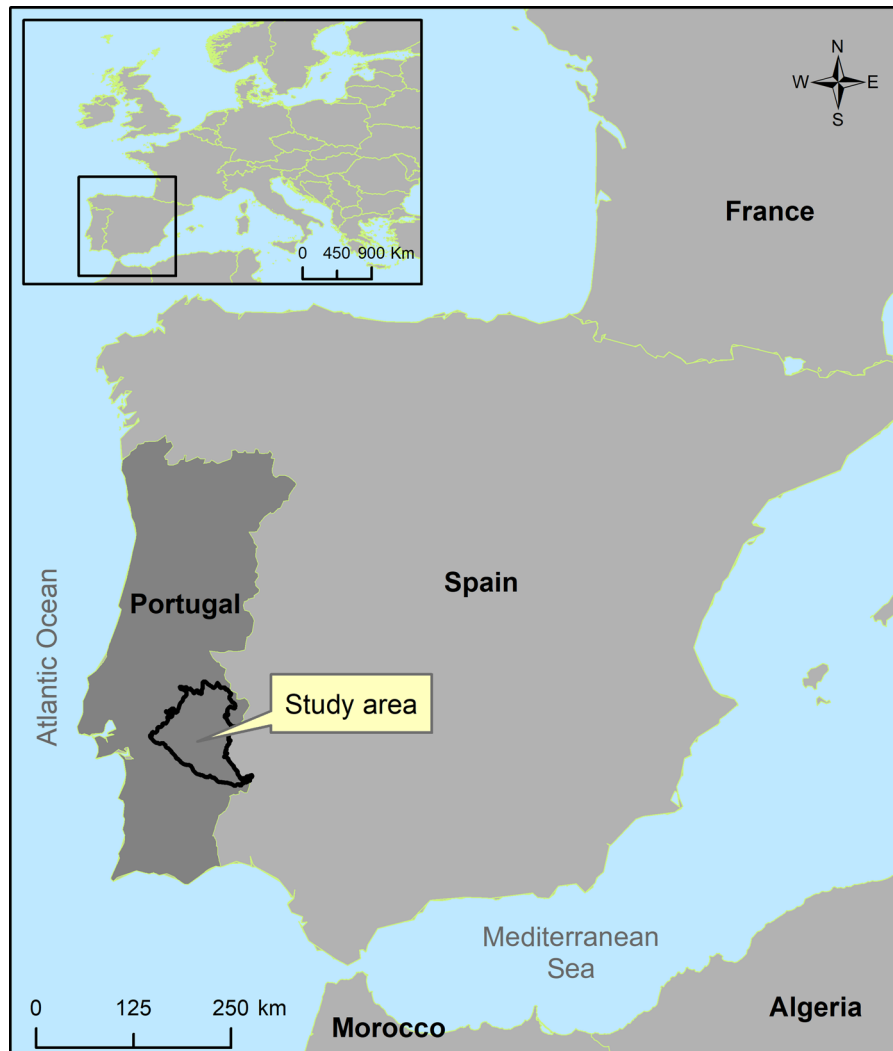


Fig. 1 Study area

same space in the landscape (Pinto-Correia 1993). The system has found economic sustainability for centuries based on a diversity of products such as cork, cereal and livestock, but also complementary products such as wool, firewood and charcoal, among others (Gaspar et al. 2007).

The climate is markedly Mediterranean characterised by hot and dry summers and wet and cold winters. Mean annual precipitation varies between 550 and 650 mm. The elevation ranges from 40 to 645 m and is characterised by soft relief. The study area, selected according to the biogeography boundaries, comprises the Alto Alentejano Superdistrict (Costa et al. 1998). *Montado* covers about 44.8% of the study

area, being the predominant land use system in the region, followed by arable land (27.9%).

Montado land cover maps production

The Landsat program provides the largest temporal records of space-based Earth observations, having been acquiring images of Earth's surface for more than 40 years (Roy et al. 2014). The spatial and spectral resolutions of the multispectral data acquired by the Landsat Thematic Mapper (TM), Enhanced Thematic Mapper Plus (ETM+), and Landsat Operational Land Imager (OLI) make it suitable for mapping and monitoring *montado* ecosystem at a 30 m spatial

resolution since the early eighties. However, producing accurate *montado* maps is a challenging task due to the fuzzy boundaries, in great part caused by tree density variability (Van Doorn and Pinto-correia 2007). Nevertheless, recent works have shown that by combining remote sensing with machine learning algorithms it is possible to produce such maps with fairly good accuracy levels (Godinho et al. 2016a, b; Allen et al. 2018). In this study, satellite imagery from Landsat TM, ETM+, and OLI sensors (path 203 and row 33) were used to produce the *montado* land cover maps for 1984, 1999, and 2014, respectively. For each year, two Landsat scenes of the same study area were acquired, one in spring and the other one in summer, to ensure that inter-class separability benefited from phenological variation of the vegetation cover (Rodríguez-Galiano and Chica-Olmo 2012; Godinho et al. 2016b). Due to the absence of cloud-free Landsat images over the entire spring season of 1984 and 1999, the images from April and June of 1985 and 2000, respectively, were used instead. As a pre-processing step an atmospheric correction was applied to the Blue, Green, Red, NIR, SWIR1 and SWIR2 Landsat bands using the FLAASH (Fast Line-of-sight Atmospheric Analysis of Spectral Hypercubes) method (FLAASH 2009). After atmospheric correction, six vegetation indices were computed for each season as auxiliary variables in image classification procedure: Enhanced Vegetation Index (EVI), Short-Wave Infrared Ratio (SWIR32), Carotenoid Reflectance Index 1 (CRI1), Green Chlorophyll Index (CIgreen), Normalised Multi-band Drought Index (NMDI), and Soil-Adjusted Total Vegetation Index (SATVI)—for more details about the effectiveness in using these indices in semi-arid environments such as the one here addressed see Godinho et al. (2016b). For each year, a layer stack with Landsat bands and derived vegetation indices was produced and classified into eleven representative land cover types: (1) *montado*, (2) pine forest, (3) eucalyptus forest, (4) olive groves, (5) vineyards, (6) irrigation agriculture, (7) dry crops/pastures, (8) shrublands, (9) water bodies, (10) bare soils, and (11) urban areas. Stochastic Gradient Boosting (SGB) algorithm (Friedman 2002) was used to classify Landsat scenes using 1300, 1301, and 1549 sample points (80% for training and 20% for validation) for the years 1984, 1999, and 2014, respectively. Sample data collection was performed based on a stratified approach by land cover types (Table 1) and through

Table 1 List of land cover types and number of sample points used for classification

Class code	Class name	Number of sample points		
		1984	1999	2014
MO	<i>Montado</i>	417	417	420
EF	Eucalyptus forest	80	83	117
SL	Shrubland	80	81	81
PF	Pine forest	80	80	80
WT	Water	90	85	89
OG	Olive grove	110	102	148
IA	Irrigation agriculture	80	81	101
C/P	Dry crops/pastures	123	117	213
BS	Bare soil	80	80	81
UB	Urban	80	80	80
VI	Vineyards	80	95	139

Landsat image-interpretation and posterior cross-validation process using different auxiliary georeferenced information; (i) the national Land Cover Map of 1990 (scale 1:25,000) in the case of 1984; (ii) the high-resolution true-color orthophotomaps (0.5 m resolution) produced in 2005 by CNIG—National Center of Geographic Information for the 1999 period; and (iii) the high-resolution imagery from 2011 to 2013 available in Google Earth for the 2014 moment. In addition, for the 1999 and 2014 periods, and taken the advantage of the panchromatic band in ETM+ and OLI sensors, a true-color composition at 15 m spatial resolution of the summer season was produced for improving the photo-interpretation process.

For the accuracy assessment of the land cover products, the validation data set was used to produce the confusion matrix and compute four accuracy measures: overall classification accuracy (OA), producer's accuracy (PA), user's accuracy (UA) and Kappa coefficient (K) (Congalton and Green 2009).

Montado connectivity assessment

In a graph theory framework, landscapes are represented as graphs and their elements correspond to graph elements. In this study, *montado* patches are considered nodes that are surrounded by the non-*montado* land cover that is less suitable or unsuitable for the *montado* species. Distances between *montado* patches associated to the species threshold

dispersal distances (d) dictate if a link exists between two nodes. A group of linked nodes forms a connected region, known as component in the graph theory terminology (Pascual-Hortal and Saura 2006). Therefore, a *montado* component encompasses the available habitat for a given specimen because it cannot reach other *montado* components.

Global connectivity

Connectivity analysis was approached via graph theory using the software “Conefor 2.6” (Saura and Torné 2009). The analysis was based on the Integral Index of Connectivity (IIC) (Pascual-Hortal and Saura 2006) that adequately detects different relevant changes in the landscape (Saura and Pascual-Hortal 2007). The IIC is given by:

$$IIC = \frac{\sum_{i=1}^n \sum_{j=1}^n \frac{a_i a_j}{1 + nl_{ij}}}{A_L^2}$$

n —total number of nodes in the landscape; a_i and a_j —attributes of nodes i and j ; nl_{ij} —number of links (shortest path) between patches i and j ; A_L —maximum landscape attribute.

When the nodes are representing habitat patches and the attribute is the area then A_L is the total area of the analysed region, habitat and non-habitat. IIC ranges between 0 (no habitat present in the study area) and 1 (all the study area covered by habitat). Although our goal was to assess how connectivity changed over time, the study did not focus on a particular species. Instead, IIC was calculated for each date using several d : 50 m, 100 m, 200 m, 300 m, 500 m, 1500 m and 2000 m. This wide range of d encompasses a large variety of important species for biodiversity conservation in the *montado* ecosystems (Sutherland et al. 2000; Martín-Martín et al. 2013), such as small and medium mammals (Rosalino et al. 2009), reptiles (Godinho et al. 2011), amphibians (Ferrand de Almeida et al. 2001) and passerines (Godinho and Rabaça 2011).

The parameter d , that is not present in the IIC expression but instead is provided to the software during the analysis, highly affects the IIC because it has direct implication in the number of links.

Individual patch importance

The relative importance of each patch to the global connectivity is given by the dIIC. The values are obtained by simulating a change in the network, such as the removal of a certain patch (Pascual-Hortal and Saura 2006). dIIC is given by:

$$dIIC = 100 \cdot \frac{IIC - IIC'}{IIC}$$

where IIC is the value of the global connectivity before the change and IIC' the value of the same index after the change. We used dIIC values to produce maps showing the different patches contribution to the global connectivity for the three dates. This task was performed using a conservative $d = 200$ m because species with moderate dispersal abilities are more likely to be affected by landscape changes related to connectivity (Saura and Rubio 2010).

IIC is constituted by three partial indices, IIC_{intra} , IIC_{flux} and $IIC_{connector}$. The first represents the intra patch connectivity, the second considers the fluxes in the landscape (inter patch connectivity) and the third calculates the importance of each patch in keeping others patches connected. Derived from these, there are $dIIC_{intra}$, $dIIC_{flux}$ and $dIIC_{connector}$, which represent the partial contribute of each patch to these primary indices (Saura and Rubio 2010).

$$dIIC = dIIC_{intra} + dIIC_{flux} + dIIC_{connector}$$

Top component analysis

A component is a patch or group of patches isolated from the other patches of the same land cover. By definition, an individual lives in one component (single or multi-patch) and is unable to reach other patches of the same habitat. In this work, the term “habitat” should be understood as equivalent to the land cover type *montado*. We calculated the components for the different dates using $d = 50$ m, 100 m, 200 m, 300 m, and 500 m, in an attempt to assess how the *montado* availability/reachability varies for species with different moving abilities. Next, we focused on the five larger components, analysed them in terms of area and number of patches (NP) and produced maps showing their spatiotemporal dynamics for $d = 200$ m.

Results and discussion

Montado land cover maps production

The performed SGB classification using multi-seasonal Landsat spectral bands and selected vegetation indices showed an overall moderate agreement and good accuracy for all the years: 1984 (OA = 81.85%; K = 0.78), 1999 (OA = 75.58%; K = 0.71), and 2014 (OA = 80.07%; K = 0.77). Regarding the *montado* areas, the SGB classification also produced reasonable accuracies: 1984 (PA = 89.2%; UA = 74.7%), 1999 (PA = 91.6%; UA = 80.0%), and 2014 (PA = 85.7%; UA = 75.8%). Based on the confusion matrix computed for each year (see Supplementary material), it can be seen that some areas of *montado* were misclassified as olive groves, vineyards and dry crops/pastures, and vice versa. As outlined in Godinho et al. (2016b), these errors occurred due to the spatial variability in tree density in *montado* ecosystem, as well as the sparse nature of the vegetation cover in olive groves and vineyards. Specifically, in low-

density *montado* areas (tree cover between 10 and 30%), olive groves and also in vineyards, the high reflectance from the understory vegetation and soils can overwhelm the reflected components of the sparse canopies contributing to a lower inter-class separability between these land cover types (e.g. Berberoglu et al. 2000, 2009; Fisher et al. 2016). In order to deal with this context-specific nature of the *montado* ecosystem, the use of high spatial, spectral and temporal resolution data such as the ones provided by the Sentinel-2A and 2B sensors may be explored (Godinho et al. 2018). Nevertheless, despite these errors in *montado* areas classification, which were in fact consistent over the three years studied, the comparison between the *montado* maps produced for each year seems acceptable considering the average classification accuracy obtained for 1984 (81.9%), 1999 (85.8%), and 2014 (80.8%). The *montado* patches were then extracted from these original land cover maps and used for the subsequent analysis (Fig. 2).

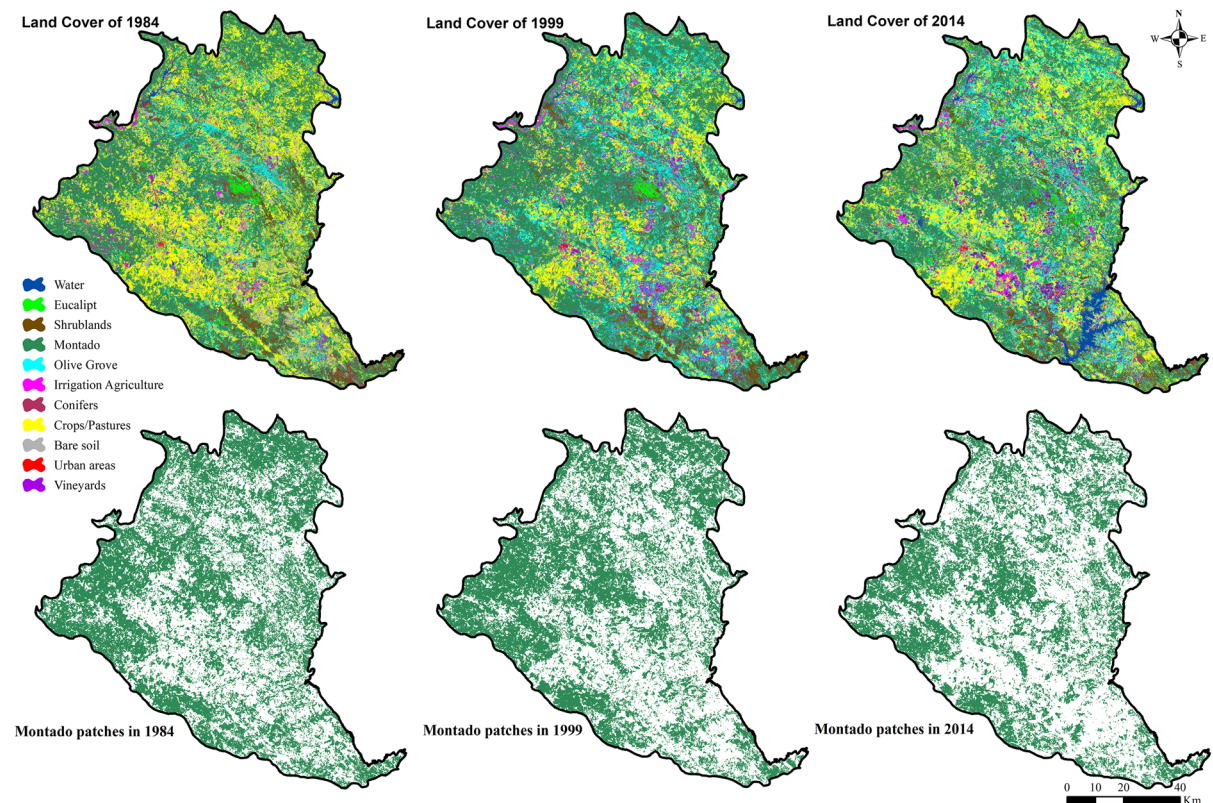


Fig. 2 Land cover and *montado* maps

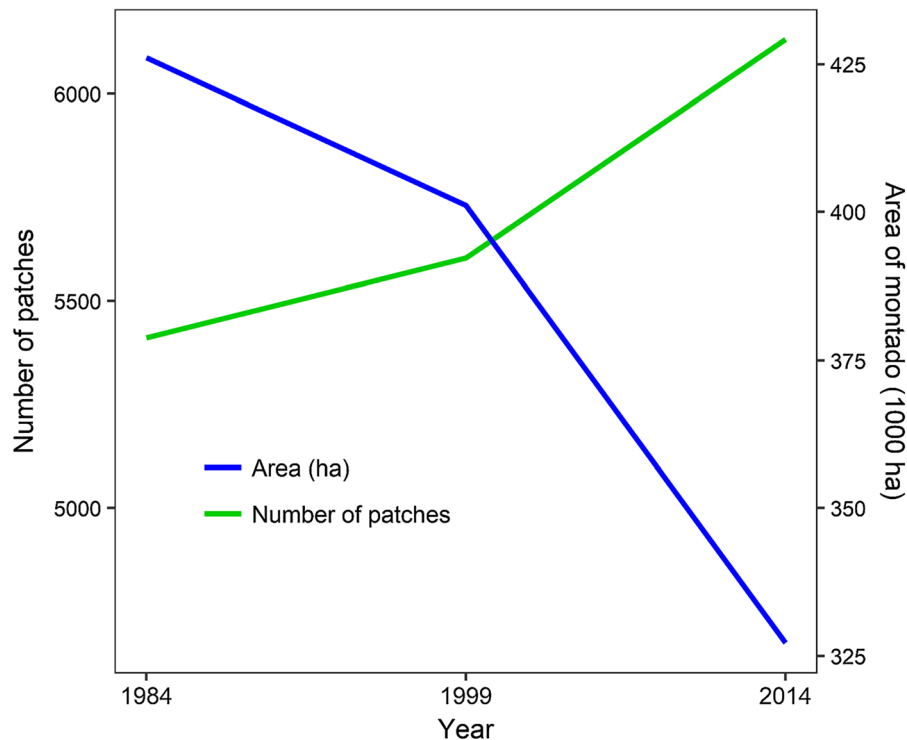


Fig. 3 *Montado* area and number of patches over time

The total *montado* area in 1984, 1999 and 2014, was respectively 426.062 ha, 401.118 ha and 327.178 ha, therefore totalizing a loss of 98.884 ha. This result is in line with the trend reported by Godinho et al. (2016c) in a study focused on *montado*, using distinct methods and a larger study area. Regarding NP, it showed an opposite trend, increasing from 5411 to 5603 to 6131, resulting in an increment of 720 patches (Fig. 3). Considering partial and overall changes, the trends are the same: *montado* area decreased and the number of *montado* patches increased. The area loss reveals a change in the *montado* composition (how much there is) and the NP increment shows a change in the *montado* configuration (how it is distributed). The combination of both suggests the existence of *montado* fragmentation due to area loss (Machado et al. 2018) at the landscape scale.

Montado connectivity analysis

Global connectivity

Montado connectivity as a whole, expressed by IIC, has been decreasing over time. The 1984 year values

are higher than those of 1999 which are higher than those of 2014 (Fig. 4). Probable reasons for “fragmentation by loss” dynamics having caused lower connectivity are (i) the *montado* loss direct influence in the IIC_{intra} and (ii) the fragmentation’s potential contribution to the loss of links, mainly for smaller d . The results also show that connectivity increases with the increment of d , which is expected because the amount of available *montado* depends on the species dispersal ability. More mobile species, able to cross non-*montado* areas to reach other *montado* patches, use a larger portion of the existing *montado* in the landscape (Saura and Rubio 2010). Results show that IIC for species with $d = 500$ m, was around four times less in 2014 (≈ 0.1) compared to 1984 (≈ 0.025). Even species with $d \approx 2000$ m suffered similar impacts in what general connectivity is concerned (Fig. 4).

Individual patch importance

Focusing on the patches individually allows us to assess which ones contribute the most to preserve connectivity (Jordán et al. 2003). The dIIC changes

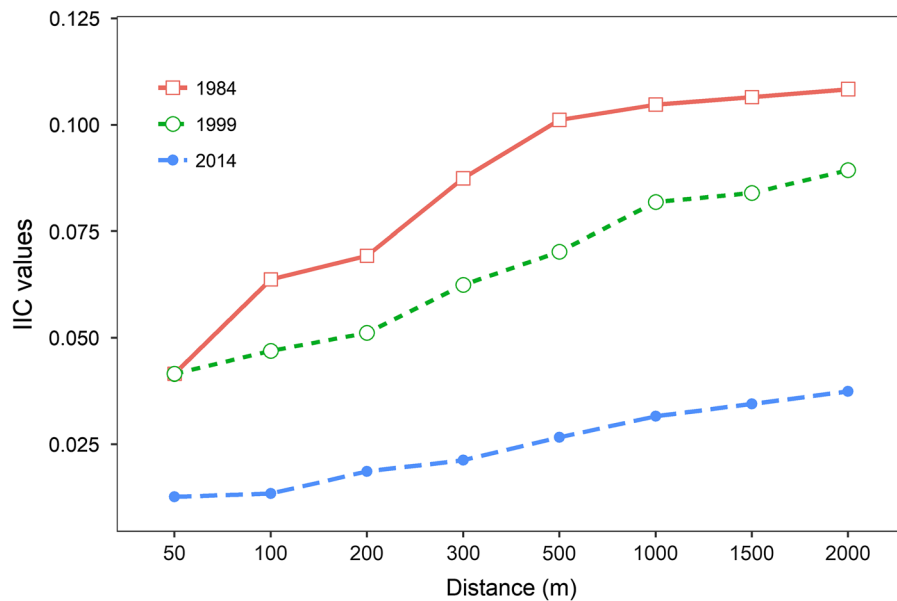


Fig. 4 Global connectivity (IIC) for different d over time

indicate the patches' relative importance to the global connectivity has varied over time. Particularly, the multi-temporal analysis using $d = 200$ m (Fig. 5) showed a noteworthy variability in the importance of some patches and makes clear that the “fragmentation by loss” occurred in the *montado* led to a reduction of its connectivity. Some resilient patches that have been fundamental to preserve the global connectivity, and kept high importance over time, have been losing area. This trend is particularly concerning given the amount of Natura 2000 sites present in the study area. See for instance the loss of importance of the *montado* in the “Monfurado” and “São Mamede” Special Areas of Conservation (PTCON0031 and PTCON0007, respectively) and “Évora” Special Protection Area (PTZPE0055).

Once a patch can contribute to connectivity with several functions and play different roles in the network (Estrada and Bodin 2008; Saura and Rubio 2010) we must go beyond $dIIC$ in order to get more in-depth information. A patch can be important due to its size (provides intra connectivity), because its location facilitates fluxes in the network (provides inter connectivity) and/or because it connects two or more patches or groups of patches that would otherwise be disconnected (act as connector). The maps in Fig. 6 show the evolution of $dIIC_{intra}$, $dIIC_{flux}$ and $dIIC_{connector}$ over time. The variations of all indices in the

years 1984, 1999 and 2014 are evident. Intra connectivity ($dIIC_{intra}$) depends only on the patch size and not on d or how the patch is connected to other patches (Saura and Rubio 2010). For that reason, it reflects directly the *montado* amount and changes in patch size. Area gain originates higher $dIIC_{intra}$ and area loss originates lower $dIIC_{intra}$. Intra connectivity showed moderate relevance in the central/western part of the study area in 1984, increased its importance until 1999, and kept high values although losing area by 2014.

Inter connectivity ($dIIC_{flux}$) depends on the patch area and on its position within the network (Saura and Rubio 2010). Therefore, changes result from both *montado* composition and configuration. This means not only the quantitative dynamics (loss and gain) of *montado* are important but also the geometric dynamics (NP increment or reduction as each patch position is relevant for establishing links/paths). Inter connectivity was very relevant in the central, western and northern zones of the study area in 1984. After, it lost some of its value in the centre and west, and most of it in the north (1999). Finally, in 2014, it is possible to see some recovery of the value but a loss of area is also noticeable.

$dIIC_{connector}$ depends exclusively on the topological position of a patch in the landscape network (Saura and Rubio 2010). Patch area is irrelevant for this index

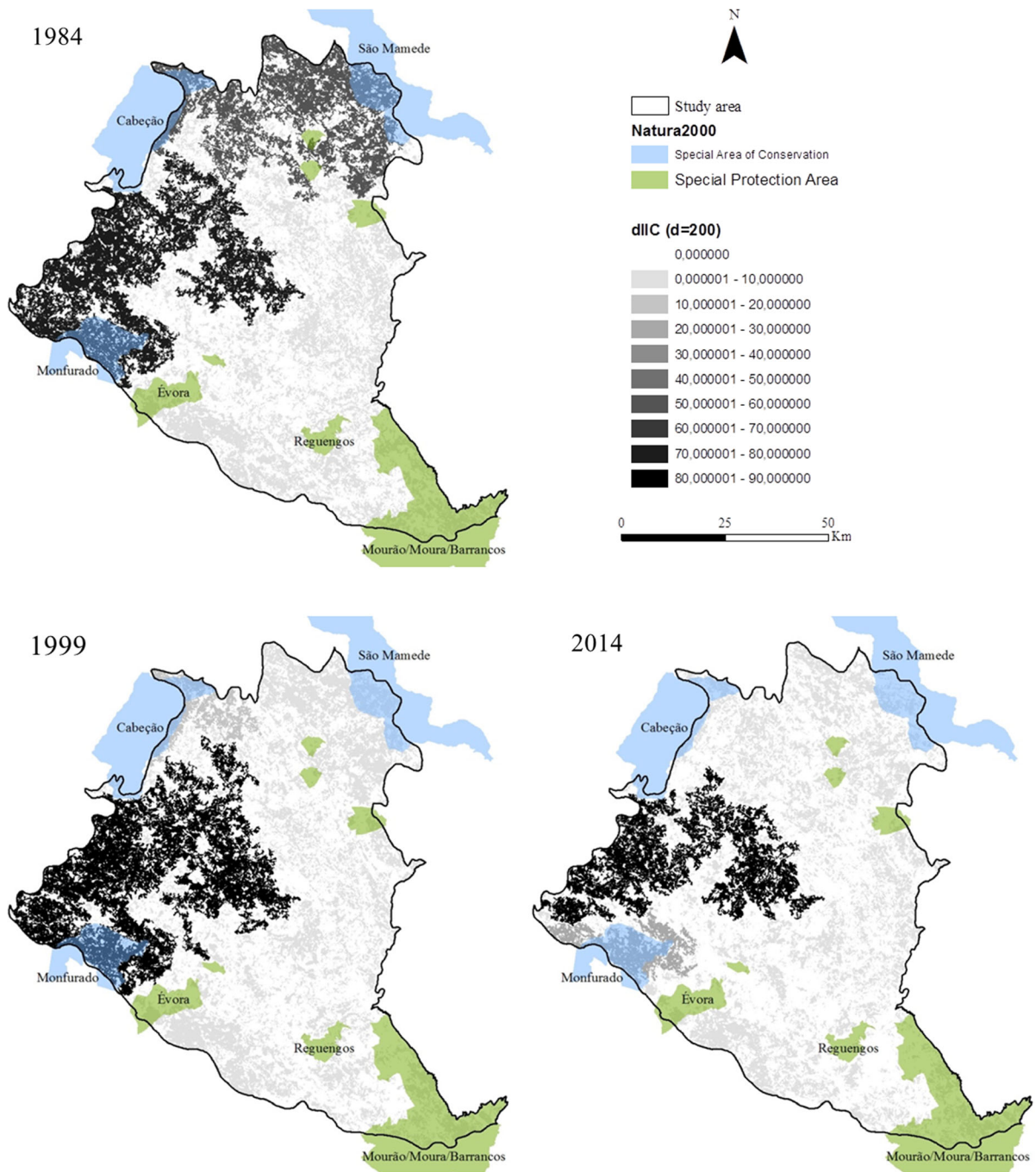


Fig. 5 dIIC maps for $d = 200$ m

that reflects only the landscape configuration dynamics. In this context, changes are consequence of increasing or decreasing *montado* NP due to the role some of them may play as stepping-stones keeping the components' cohesion. $dIIC_{connector}$ is the fraction

with the most variations of all three (Fig. 6). In 1984, it was highly important in the northern part of the study area and moderately important in the centre and west. In 1999, it had lost most of the importance in the north and kept stable in the centre/west. By 2014, it had

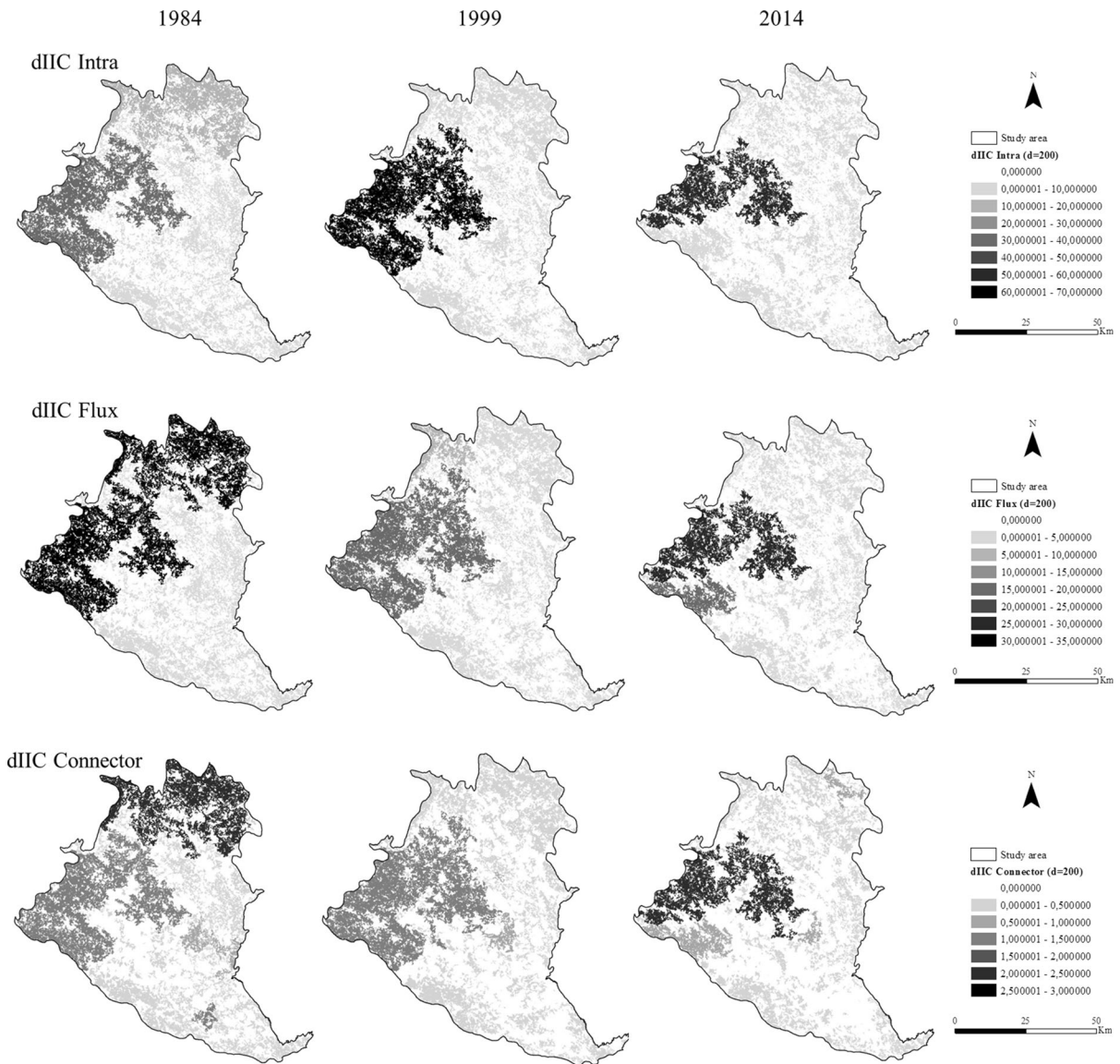


Fig. 6 $dIIC_{intra}$, $dIIC_{flux}$ and $dIIC_{connector}$ maps for a $d = 200$ m

increased its value although its area decreased. These variations are expected to have significant impacts in the populations because the connector patches are directly implied in habitat availability as their existence or inexistence determines more or less habitat at disposal (Gurrutxaga et al. 2011). In a scenario of habitat loss and increasing fragmentation, the connectivity between patches becomes even more important because it may be the only way in which enough habitat can be made available for populations (Jordán 2001).

The importance of knowing the contribute of each patch to the global connectivity is clearly demonstrated in the work by Dondina et al. (2018). Their study aimed at assessing connectivity in an agroecosystem in the Lombardy region (Italy) and was focused on the hazel dormouse. By comparing actual and simulated land covers, the authors found that establishing hedgerows along new pathways would promote higher connectivity gains than restoring/improving existing corridors. In that particular case, instead of trying to enhance relatively less important

elements (with low dIIC, for instance), it would be more beneficial to promote new ones.

Top component analysis

While dIIC analysis works at the patch level and requires deeper understanding of the underlying processes, landscape components analysis is a more expedite and simple way to extract valuable information (Neel et al. 2014). As previously mentioned a component is a connected region and represents the maximum habitat available for a given individual (Pascual-Hortal and Saura 2006). Larger components are expected to harbour more species than smaller ones as they provide more suitable habitat, are less prone to stochastic extinctions and thus are more likely to maintain viable populations (MacArthur and Wilson 1967; Lande 1988). Besides the size, the number of components (NC) also matters because it is directly related to connectivity and habitat availability. For a given area, more components are synonym of discontinuity and less available habitat.

It is important to clarify that NC itself, is not good or bad. For instance, a *montado* area gain located far away so that it would remain disconnected from the network, (e.g. a new plantation) originating a new patch, would form a new component that would be understood as beneficial. However, in our case study we know the *montado* area has been decreasing and the dynamics occurring is “fragmentation by loss” rather than “number of patch increment by gain” (Godinho et al. 2016c; Machado et al. 2018).

The numbers of components for different d in each moment are shown in Fig. 7a. Naturally, the values decrease as the d increase because species with a larger d can reach more patches than a species with a smaller d . For the former, there will be fewer components in the landscape because there are fewer unreachable patches. Conversely, for the latter there are more unreachable patches and thus, more components. The values of each d increased over time reflecting the increasing number of isolated patches and consequent lower connectivity. This also means that many components are composed by a single patch with virtually no influence in the *montado* connectivity.

The overall *montado* area reduction is noticeable when we look at the five larger components. These constitute the core structure of the *montado* and comprise most of the available area for species with

$d > 200$ m (Fig. 7b). For lower d (50 m and 100 m) this is also verifiable for 1984 and 1999. By 2014, changes in the *montado* had been so severe that more components were needed to comprise a higher percentage of *montado* area. For higher d (300 m and 500 m) the values remain similar along the years because the species are mobile enough to reach *montado* patches despite the loss of area occurred. This means the majority of the remaining *montado* patches are closer than 300 m.

A comparable analysis considering the NP percentage instead of area percentage has similar results showing that for a given d , the NP accessible has been decreasing over time (Fig. 7c). Species capable of movements of 300 m and higher have access to almost all the *montado* patches in the landscape while species with very limited moving abilities (50 m or 100 m) only have access to a small percentage of patches.

A more detailed analysis for $d = 200$ m shows how the available *montado* area has decreased over time. It's also noticeable the five larger components have experienced little stability during the studied period (Fig. 7d). Between 1984 and 1999, the top 5 components lost 68,812 ha (401,362–332,550 ha). Between 1999 and 2014, the top 5 components lost 104,424 ha (332,550–228,126 ha). Between 1984 and 2014, the top 5 components lost a total of 173,236 ha (401,362–228,126 ha) (Fig. 7c).

The maps in Fig. 8 show how the five largest components changed during the studied period for $d = 200$ m. Taking in consideration that one individual can only move within one component it is clear the habitat has become more fragmented and less accessible. For instance, a population living in the largest (green) component in 1984 had its component split in two by 1999 (green and yellow), which means those patches were no longer linked and consequently the population was divided.

Which species or group of species are the most affected by the increasing number of *montado* components?

First of all, it is important to stress the *montado* biological value does not come from a single species, nor there are any exclusive species to this land use system. Instead, its richness in terms of biodiversity results from the quantity of species it harbours. Thus, changes in the *montado*, such as splitting components, more than affecting a single species, are expected to provoke changes in the species pool. Any species

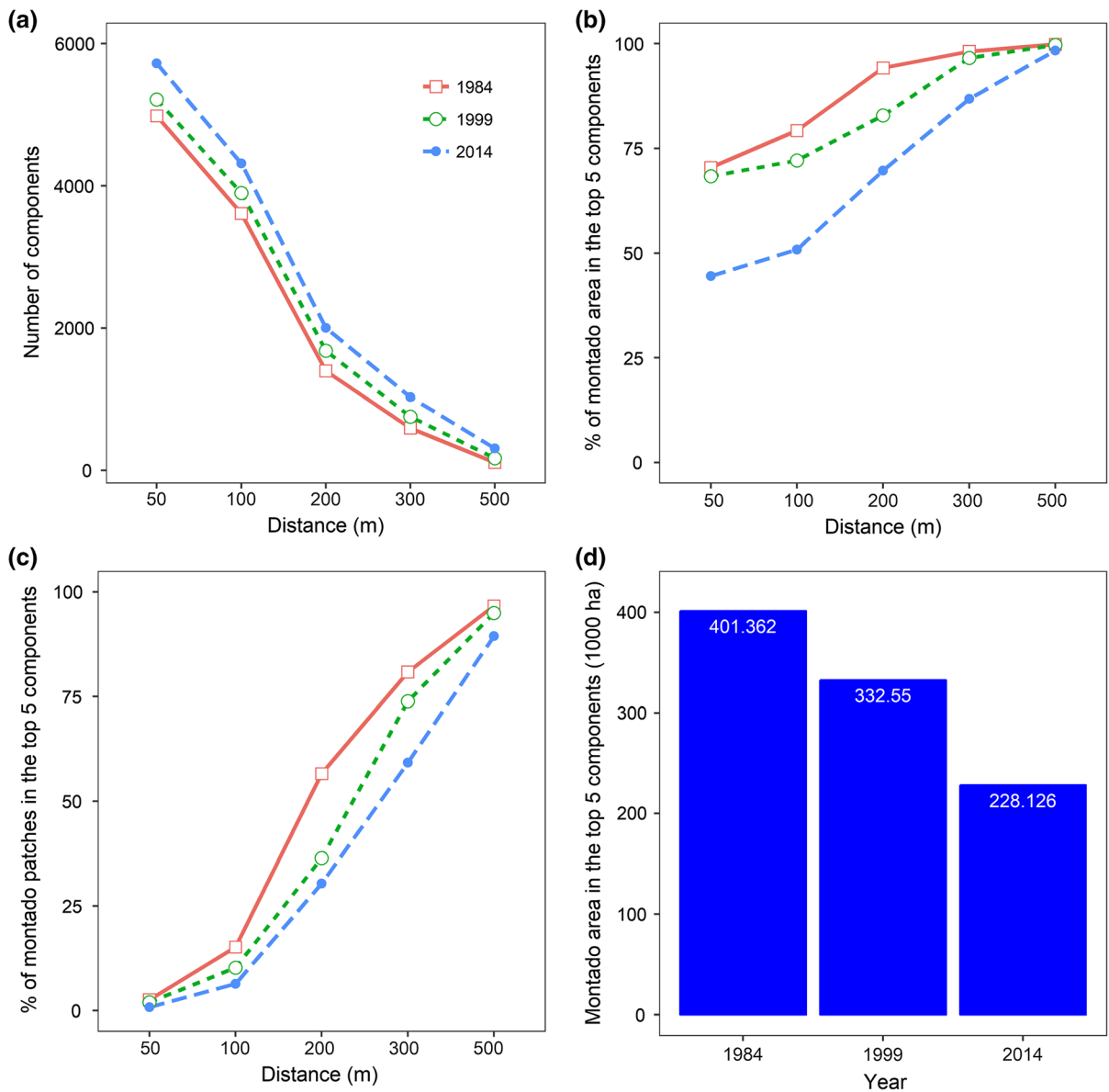


Fig. 7 Montado components in 1984, 1999 and 2014; **a** Number of components for different dispersal distances (d); **b** Montado area (%) in the top 5 components for different d ; **c** Montado

patches (%) in the top 5 components for different d ; **d** Montado area in the top 5 components for $d = 200$ m

whose habitat is reduced and disconnected is affected to some extent. In this particular analysis the resulting components show how fragmented the *montado* has become for species with $d \leq 500$ m (Fig. 7). Many small mammals belong to these group, among them the *Apodemus sylvaticus* (Dickman and Doncaster (1989) that favours oak recruitment by spreads acorns (Muñoz and Bonal 2007).

Many species live in the *montado* because the system fits their habitat requirements but do not necessarily depend on it. Others, more specialized and less adaptable display a higher degree of dependence. Focusing on birds as an example, farmland specialists and forest specialists occur only in certain farmlands and forests, respectively. Common farmland birds can occur in a variety of open lands and the common forest birds can live in several forest types. There are also

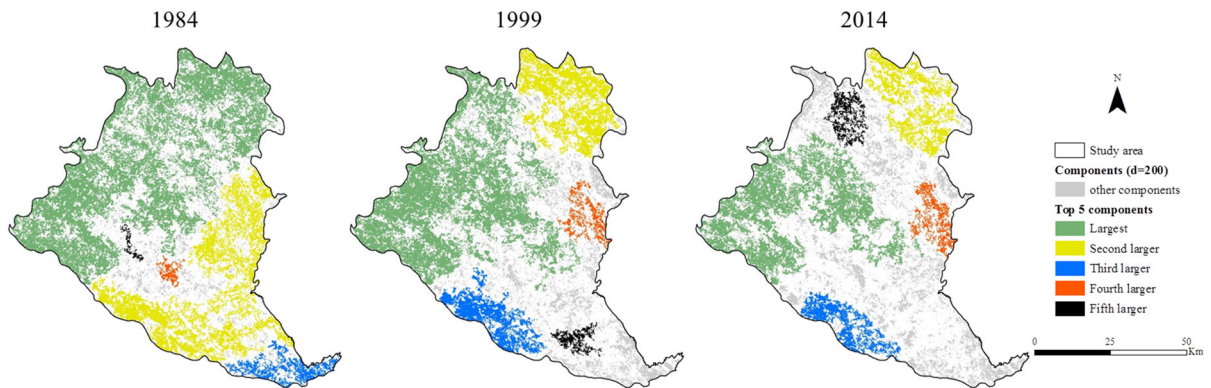


Fig. 8 Spatiotemporal dynamics of the *montado* top 5 components for $d = 200$ m. Larger components by size: 1st—green, 2nd—yellow, 3rd—blue, 4th—red, 5th—black, remaining components—light grey

transition habitat birds that thrive in the interface of open lands and forests. The *montado* structure with pastures and different shrub and tree densities makes it a land use type often used by all of the above groups, except some farmland specialists that completely avoid the presence of trees (Pereira and Godinho 2015; Catarino et al. 2016). Although *montado* degradation can be harmful to many, the forest specialists who live in the *montado* are the most dependent on the system stability and thus are the most affected by a level of loss and/or fragmentation that breaks *montado* components apart. In our study area, *Sitta europaea* and *Dryobates minor* are examples of species well adapted to mature *montados* that are likely to be more affected by the increasing NC.

Conclusion

The *montado* classification as high nature value farming system helped to raise awareness to its importance but the heterogeneity that differentiates the *montado* has not been reflected in the policies. Several policy instruments affect the *montado* and have allowed its economic valorisation but do not consider it as a whole, compromising the survival of many *montados* in the medium-term (Pinto-Correia et al. 2013). Concrete measures are needed that effectively improve multi-functional farming systems and support practices that preserve biodiversity such as extensive grazing in high nature value farmlands (Pe'er et al. 2020).

The biodiversity aspect is particularly relevant in the *montado* because the system's well-functioning

and sustainability is highly dependent on natural processes. Without proper biological support, *montado* becomes a poor and unsustainable system threatened by abandonment. Thus, it is essential to guarantee suitable conditions for wildlife to prosper and for that connectivity plays a key role.

Montado changes and its resulting connectivity in the years 1984, 1999 and 2014 were assessed based on accurate information derived from satellite remote sensing multispectral data. In this period:

- *Montado* experienced a deep regression in terms of area amount (− 98.884 ha) and an increase in the fragmentation level (+ 720 patches).
- The global connectivity decreased for all the tested d , resulting in a negative impact for several species.
- The relative importance of each *montado* patch to the global connectivity varied over time due to changes in spatial composition and configuration.
- The *montado* area represented by the top 5 components varied significantly and the amount displays a decreasing trend. This implies that species with dispersal abilities up to 500 m in 1984 had more *montado* connected areas than in 1999 and 2014. Ultimately, it means the populations are increasingly more isolated in 1999 and 2014.

The most affected species by habitat loss and/or fragmentation are those with low mobility that cannot reach other viable patches via dispersal and those that display higher levels of specialization and thus are more dependent on the habitat's characteristics. In the *montado*, the former are mainly micromammals and

the latter some forest specialist birds such as the *Sitta europaea* and the *Dryobates minor*.

Every *montado* patch (and component) is important but the larger ones are more likely to display higher specific richness, seem to be more resilient and provide more intraconnectivity that is key to short dispersal species, unable or unlikely to move successfully among suitable patches in a fragmented landscape. Hence, at the landscape scale, the priority should be to preserve the core areas that have been the support of connectivity (the larger components) and avoid as much as possible the loss of more area that will consequently provoke fragmentation. A more fragmented *montado* (higher NC) can also mean that more patches need to be preserved to guarantee high habitat availability and therefore making conservation prioritization harder (Pereira 2018). Biodiversitywise, *montado* has value as a whole and given also its socio-economic importance measures should be taken in order to preserve it as much as possible. By reporting the *montado* decreasing trend and highlighting some of the associated impacts on biodiversity, on which the *montado* itself depends upon, our work also contributes to raise awareness and reinforce the need to preserve and increment this unique land use system.

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