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Temporal grazing exclusion as a passive restoration strategy in a dryland woodland: effects over time on tree regeneration and on the shrub community

Authors: Köbel, M[a]; Listopad, C[a,b]; Príncipe, A[a]; Nunes, A[a]*; Branquinho, C[a]

[a] cE3c, Centre for Ecology, Evolution and Environmental Changes, Faculdade de Ciências da Universidade de Lisboa, Campo Grande, edifício C2, 5º piso, Lisboa 1749-016, Portugal

[b] Florida Institute of Technology, Department of Ocean Engineering and Sciences, 150 W. University Blvd., Melbourne, FL 32901, USA

* Corresponding author

MK: mkobel@fc.ul.pt
CL: clistopad@appliedecologyinc.com
AP: aprsilva@fc.ul.pt
AN: amanunes@fc.ul.pt
CB: cmbranquinho@fc.ul.pt

Abstract

Passive restoration is acknowledged as a cost-effective approach to restore forest ecosystems. In this approach, restoration implies simply ceasing the disturbance causing degradation. But an efficient allocation of restoration efforts requires an understanding of the recovery trajectories of different ecosystem components.

In Mediterranean oak open woodlands (called *montado* in Portugal), livestock grazing drives important plant community composition changes, and may hamper tree regeneration, threatening the long-term sustainability of this ecosystem. A temporsl grazing exclusion may, thus, be an efficient restoration measure. However, how beneficial changes occur with time after exclusion on tree regeneration and on the structure and diversity of the ecosystem, are less clear.

In this work, we studied the effects of grazing exclusion as a passive restoration strategy on a cork oak woodland, focusing on cork oak tree (*Quercus suber*) regeneration and on the structure and diversity of the understory shrub community. We assessed juvenile oaks’ abundance and height, shrub structure, compositional and functional diversity at sites with 0, 5, 9 and 13 years of grazing exclusion.

Cork oak individuals were similarly abundant throughout the chronosequence, but time had a positive effect on the height of saplings, until 9 years of exclusion. Shrub cover increased within 5 years after exclusion, and remained stable thereafter. Shrub taxonomic diversity increased continuously along the chronosequence, while functional diversity did not change consistently.

Our results suggest that grazing exclusion may be an effective passive restoration measure, although different components of the ecosystem have different timings of recovery. In the case of the studied cork oak woodlands, tree regeneration may be promoted by a 5-10 years period of exclusion, after which it decreases, while conservation of taxonomic shrub diversitymay require at least 13 years of exclusion. Our results highlight the importance of optimizing restoration plans depending on the restoration objective.

Keywords

Cow grazing; Cork oak; High Nature Value Farmland; Mediterranean; *Quercus suber;* Secondary succession.

1. Introduction

The importance of ecological restoration increases at the rate of global ecosystem degradation (Gann et al., 2019). It consists on actions applied to a degraded ecosystem, with the aim of assisting its recovery to a historical trajectory, or recovery of its biodiversity, functions, or provision of valued ecosystem services (Martin, 2017). two main approaches are broadly recognized: i) Passive restoration – ending the disturbance at a given site and letting natural succession take place; ii) Active restoration – actions such as plantations, application of soil amendments or species introduction (Jones et al., 2018). Studies are increasingly supporting the need to consider ecosystem functions and services in restoration goals (Nunes et al., 2016). However, restoring ecosystem’s diversity and functions, such as species richness and soil organic matter, can be seen as a continuum evolving at different rates through time, in response to environmental conditions and feedback cycles that may occur (Jones et al., 2018). Thus, an efficient allocation of restoration efforts requires an understanding of how, and when, do ecosystems regain functions and properties that were lost.

In Mediterranean drylands, extensive areas are occupied with open oak woodlands (more than 3.5 Mha in Europe [Olea et al., 2005] , and, in 2006, ca. 1.22 Mha in Portugal alone [Godinho et al., 2016]). This ecosystem, called *montado* in Portugal, or *dehesa* in Spain, is dominated by cork oak (*Quercus suber L.*) or holm oak (*Quercus rotundifolia Lam.*), with an understory of grasslands, with species such as *Vulpia geniculata* L. Link and *Lolium rigidum* Gaudim. and shrubs such as *Cistus* spp. Traditionally, cork oak woodland land-owners combine livestock raising with cork harvesting, among other activities (Díaz, Campos, & Pulido, 1997). *Montado* areas are thus an important source of provisioning services as well as regulating and supporting services in dryland areas, such as climate and water regulation, fire risk reduction and biodiversity conservation (Bugalho et al., 2011a). Because of its high diversity, including endangered species such as the plant species Narcissus, *Narcissus fernandesii* Pedro, or the Imperial eagle, *Aquila adalberti* C. L. *Brehm*, this ecosystem is protected at the European level in the Habitats Directive 92/43/EEC (European Comission, 2013). As an European agricultural landscape that sustains high diversity of species and habitats, it is also recognized as a High Nature Value Farmland (Paracchini et al., 2008).

Over the past decades, however, a consensus has been established about a large-scale decline of the *montado* ecosystem in terms of tree cover and total area occupied: during the period 1990-2006, the total area, including both cork-oak and holm-oak dominated areas, decreased at a mean rate of 0.14% year^{-1} (Godinho et al., 2016). The causes may include pests and diseases (Camilo-Alves et al., 2013), intense management practices (Arosa et al., 2015), grazing intensification, by cows in particular (Perea et al., 2016), local unfavourable microclimatic conditions (Príncipe et al., 2014), and tree recruitment failure under species-poor shublands (Acácio and Holmgren, 2014).

In semi-natural ecosystems such as these Mediterranean oak woodlands, ecological restoration aims particularly at maintaining the biodiversity, traditional management practices and ecosystem services these systems provide, taking into account that human inputs are an integral part of the system (Gann et al., 2019). In the montado ecosystem, grazing is a traditional land-use (Bugalho et al., 2011b). It reduces above-ground plant biomass and may alter plant species composition, contributing to the maintenance of a savanna-like structure, promoting overall diversity and reducing wildfire risk (Castro and Freitas, 2009). However, even moderate grazing and browsing, over the long-term, may have negative impacts in the *montado* ecosystem, particularly contributing to a low tree regeneration, and low sustainability and resilience of this system (Godinho et al., 2016; Plieninger et al., 2003).

On the other hand, with decreasing livestock pressure a development of shrubs might be expected (Bugalho et al., 2011a; Listopad et al., 2018). Some studies report a nurse effect of shrubs, through shading and protection from herbivory (Arosa et al., 2015; Ramírez and Díaz, 2008), whereas others show that shrublands may promote acorn predators and drought-induced seedling mortality through competition (Acácio and Holmgren, 2014).

Besides the tree regeneration, grazing and browsing also alters vegetation structure, diversity and plant functional traits (Díaz et al., 2007; Peco et al., 2005), with cascading effects on animal communities and ecosystem functioning (Bugalho et al., 2011b; Castro and Freitas, 2009; Maestre et al., 2016). Assessing functional diversity of oak woodlands may, thus, be relevant for the maintenance of the multiple services these ecosystems provide. Furthermore, functional diversity drives ecosystem resilience in face of environmental change, as it reflects the multitude of strategies co-existing in the community (Folke et al., 2004). In the context of oak woodlands, this may be particularly relevant given, for example, the increasing aridity foreseen for Mediterranean drylands where these systems occur (Nunes et al., 2017). The rate of change of species and functional traits after grazing exclusion, i.e. the time needed to recover ecosystem functions and components central to *montado*’s sustainability and nature value, such as regeneration, structure and diversity, may depend on macroclimate, fire and management history (Acácio and Holmgren, 2014; Godinho et al., 2016). Thus, even when the relationship between grazing intensity and ecosystem recovery is established (Bugalho et al., 2011b), the time scale at which it recovers is seldom studied for this ecosystem (but see Cruz-Alonso et al. [2019] and Ramírez and Díaz [2008] for approaches on the close relative holm oak). To contribute to clarify this question, we evaluated the effects of grazing exclusion as a passive restoration measure on a Mediterranean open oak woodland. Along a 13-year chronosequence of grazing exclusion, we assessed tree regeneration and different components of the shrub community, namely its structure, and taxonomic and functional diversity, to evaluate how these different components recover along time since exclusion in a cork oak woodland. We will focus on the shrub community since this layer is deeply influenced by management and grazing (Castro and Freitas, 2009; Tárrega et al., 2009). This study answers two main questions: 1) What is the effect of grazing exclusion over time on cork oak regeneration? 2) How do the shrub structure, taxonomic and functional diversity change with time after grazing exclusion? Our hypothesis is that tree regeneration will be promoted by grazing exclusion, and that shrub taxonomic and functional diversity will increase with time after exclusion.

2. Methods

2.1. Study site

This study was conducted in central Portugal (38º 52'.780 N, 08º 51'.747 W) at the state-owned farm *Companhia das Lezírias.* It comprises agricultural and forestry land-uses, but the largest portion (6700 ha) is devoted to cork oak open woodlands, where the farm develops its main activities - cork extraction and cattle raising. The study area emcompasses the most cork productive region at the national level, and the farm was also established as a Long-Term Socio-Ecological Research site for the *Montado* platform, promoting long-term research and data exchange (LTsER montados, 2019).

The climate is Mediterranean, with mild wet winters and dry hot summers, with a mean annual precipitation and temperature of 608 mm, 16 ℃.

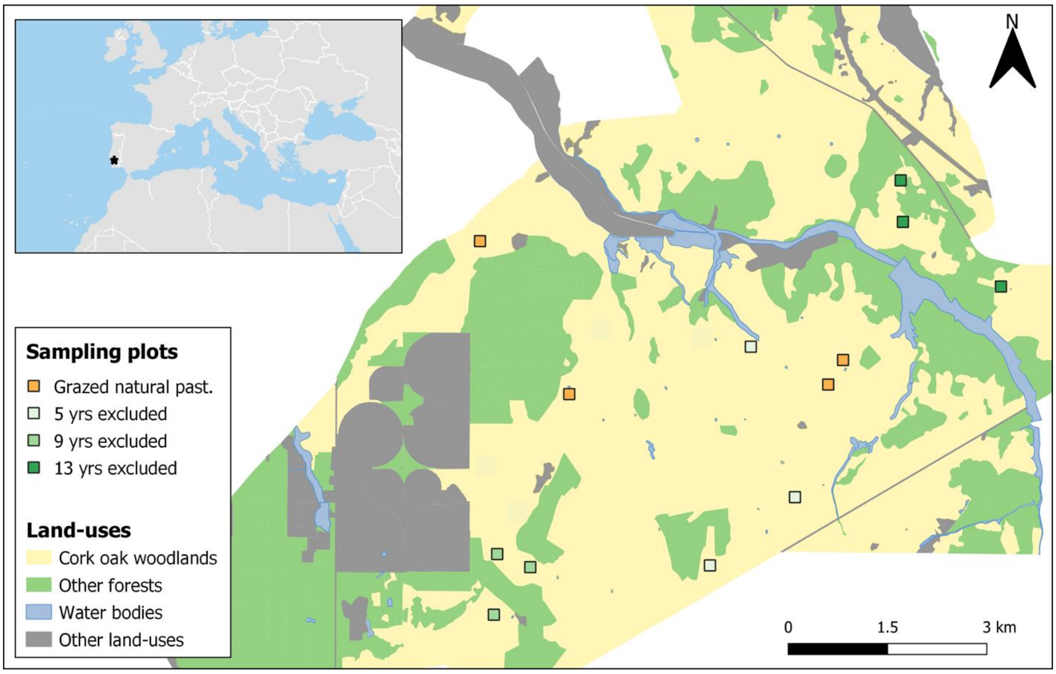

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Figure 1: Top left: Study site (star) in Portugal. Main map: sampling plots throughout the *montado* ecosystem at the LTsER site *Companhia das Lezírias*.

To promote multiple functions and uses in the farm’s woodlands, some areas were fenced to contend livestock, while in others grazing was excluded. In this work, we took this opportunity to study the effects of grazing exclusion over time on tree regeneration and on the structure and diversity of the understory shrub community. We sampled 13 quadrat plots with 1 ha, comprising 4 treatments: grazed with semi-natural pastures (4 plots), and excluded from livestock for 5 years, for 9 years, and for 13 years (3 plots each). Plots were established within larger commonly managed areas and trying to minimize confounding factors: in flat areas, at least 50 m away from other land-uses, and within similar soil characteristics. These woodlands are dominated by cork oak trees, with a whole-farm average density of 85 trees/ha (LTsER montados, 2019). Among plots, tree cover varies between 41-80%, but this variability is transversal to all ages of exclusion (Listopad et al., 2018). Maritime pine' (*Pinus pinaster*) saplings occur in excluded plots, dispersing from surrounding plantations, but are periodically cut to maintain the cork oak woodlands. In the 13-year excluded plots, some adult pines are also present, but the overstory is dominated by cork oak.

In grazed cork oak woodlands, grazing intensity varies between 0.5 and 1.1 LSU/ha (Gonçalves et al., 2012). In the semi-natural pastures, the plant community is dominated by annual forbs and grasses (e.g. *Tolpis barbata* and *Lolium rigidum).* Cork is harvested every 9 years, implying a localized shrub clearing around adult cork oaks, however this is done equally in all plots.

2.2. Tree regeneration, shrub structure and taxonomic diversity

Understory woody vegetation was sampled in early summer of 2013 and 2015, using the line-intercept method (Elzinga et al., 1998). In each 1 ha plot, four transects of 50 m were laid systematically, totaling a sample of 200 m per plot. All woody understory individuals (i.e. shrubs and cork oaks, from seedlings to saplings up to 3 m) intercepted by the transects were identified to the species level, and its cover measured by registering the length of the canopy projected along the transect, and the maximum height determined with a measuring tape (Pérez-Harguindeguy et al., 2013). With the collected data we computed shrub diversity, cover, and mean shrub height per plot. Cork oak individuals were included here as being part of the shrub layer. For assessing oak regeneration, we computed cork oak abundance and height, considering two separated size classes: below and above 0.5 m of maximum height, to differentiate seedlings from saplings with higher potential of establishment (Pons and Pausas, 2006).

Three measures of taxonomic diversity were calculated: shrub species richness (number of species recorded in each plot), diversity (Diversity of order one, ^{1}D [Hill, 1973]) and evenness (the quotient of ^{1}D by species richness). The Hill numbers, acknowledged as “true diversity” measures, calculate the effective number of species of a community, i.e. the number of equally abundant species a community can withhold, at the mean proportional abundance observed (Tuomisto, 2010). ^{1}D was computed with *renyi* function from *Vegan* package (Oksanen et al., 2019) in *R* (R Core Team, 2019).

2.3. Functional diversity

To assess the effects of grazing exclusion on shrub functional diversity, each species was characterized concerning three traits: height, leaf area and seed mass (Table A.1). Taken together, these traits are meaningful to explain species strategies and distribution (Westoby et al., 2002), composing the leaf-height-seed strategy scheme proposed by Westoby (1998), with leaf area being here used instead of specific leaf area due to data availability.

Trait values were retrieved from databases (Paula et al., 2009; Royal Botanic Gardens Kew, 2008; Tavşanoğlu and Pausas, 2018), except for height, which was measured *in situ*, as explained in section 2.2. For the computation of functional diversity, only the height of the tallest individual was used, because we were interested in the fully-grown species potential. Leaf area was included as size classes, because continuous values were unavailable for most species. Still, when trait data for seed mass or leaf area of a particular species were not available, data from congeneric species (if identical among them) were assumed for the species of interest. This, however, only happened in 9 out of 57 values (see the marked cases in table A.1).

With the values of these three traits, we calculated three functional diversity metrics.: functional richness (FRic) and evenness (FEve, both from Villéger et al. [2008]) and functional dispersion (FDis, from Laliberté and Legendre([2010]) using the *FD* package (Laliberté et al., 2014). Taken together, these different metrics compose a meaningfull framework for the measurement of functional diversity (Laliberté and Legendre, 2010). FRic measures the range of functional identities in a community. When traits values are represented in a multivariate space (in our case, we are using three traits, so it is a 3-dimensional space), FRic corresponds to the minimum volume needed to enclose all trait values occurring in the community (Villéger et al., 2008). FDis is an abundance-weighted measure of dispersion (Laliberté and Tylianakis, 2012). It is computed as the mean distance of individual species to a centroid in the multivariate functional space, weighted by species relative abundances (Laliberté and Legendre, 2010). FEve measures the evenness of species distributions along the multidimensional trait space, and is calculated by firstly drawing the minimum spanning tree linking all species in the functional space, and then quantifying how even species and its abundances are distributed along this tree (Villéger et al., 2008). To compute these indexes, the Gower dissimilarity measure was used because it accommodates different types of traits (qualitative, quantitative and semi-quantitative) (Laliberté and Legendre, 2010). Prior to these calculations, traits were standardized, height and seed mass were log transformed, and the Pearson correlation between traits was checked. Since height was correlated with leaf area, these two traits shared a weight unit in the calculations (0.5 each).

Additionally, for each of the three traits we computed single trait functional dispersion, FDis, and the community-weighted mean, CWM. Single trait FDis is similar as for the multi-trait approach. CWM, for continuous and semi-quantitative traits, corresponds to the average trait value for the whole community, weighted by species relative abundances (Garnier et al., 2007). This metric, is a measure of an aggregated trait response and reflects the dominant trait response to an environmental driver, at the whole community level (Garnier et al., 2007).

2.4. Data analysis

The effect of time since grazing exclusion on the plant community was tested using Generalized Linear Models, with vegetation-derived metrics as dependent variables, and time since grazing exclusion as the fixed factor. For abundance of oak individuals and shrub species richness, GLMs were fit with Poisson distribution (logarithmic link function), considering a dispersion parameter in case of overdispersion. If needed, variables were log transformed to improve normality of residuals, or, in case of heterocedasticity of errors, the model was fit using generalized least squares, accounting for unequal variances between periods of exclusion (Pinheiro et al., 2019). We assessed all models for temporal and spatial dependency, respectively, by testing if a correlation structure or order 1 (AR1) would improve model fit and by calculating the Moran’s I Autocorrelation Index (Gittleman and Kot, 1990), using the ape package (Paradis and Schliep, 2018). This index estimates how observations are related, based on their geographic locations. No signs of temporal or spatial correlation were found. The correlation between species richness and functional richness was tested with Pearson's correlation test. All analyses were performed in *R* software (R Core Team, 2019), and assuming an $\alpha = 0.05$ for significant differences.

3. Results

3.1. Tree regeneration

Abundance of oak individuals per plot (i.e. along four 200 m transects) ranged between 6 and 39. The abundance of individuals did not change significantly with increasing time since exclusion ($\chi^2 = 3.11$, $P = 0.416$, Fig. 2A). Similarly, no difference was found for abundance of seedlings (height below 0.5 m) or saplings (height above 0.5 m) ($\chi^2_{1,11} = 5.80$, $P = 0.306$, $\chi^2_{1,11} = 0.59$, $P = 0.692$, respectively; data not shown). However, the height of saplings showed a unimodal pattern along the chronosequence ($F_{2,49} = 2.76$, $P = 0.003$; Fig. 2B). The height of individuals increases from grazed plots (mean height 0.85 ± 0.38 m) to 9-years excluded plots (1.64 ± 0.55 m), after which it decreases in the 13-years excluded plots (0.84 ± 0.51 m).

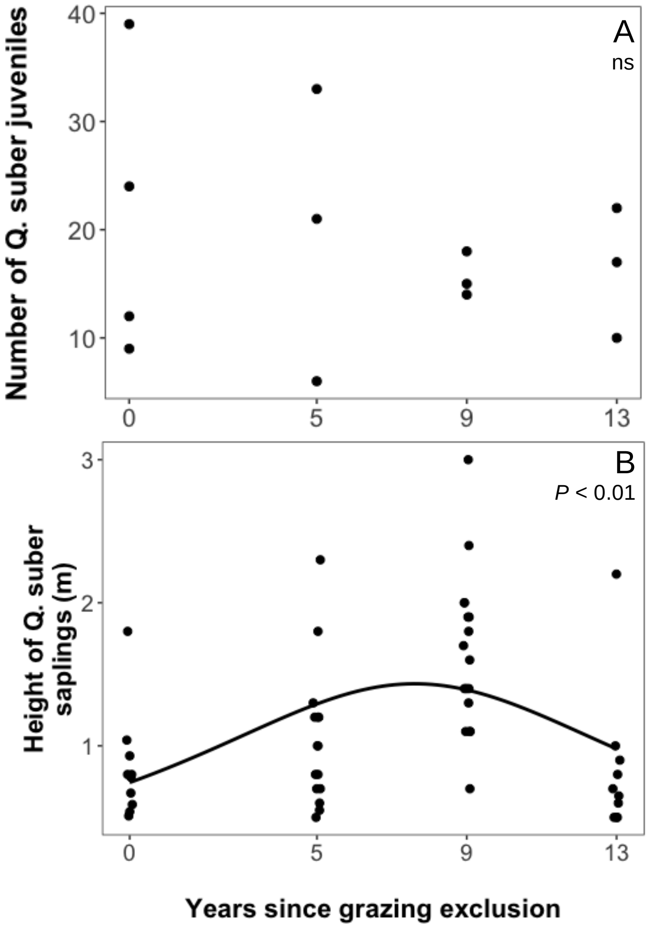

Figure 2: Cork oak individuals found in the understory. (A) abundance, showing no pattern along the chronosequence; (B) height of saplings (i.e. individuals taller than 0.5 m) showing a significant unimodal pattern with increasing age of exclusion. Here points are jittered to avoid overplotting. $N = 13$ for abundance, $N = 52$ for height, at $\alpha = 0.05$.

3.2. Shrub structure and taxonomic diversity

A total of 29 shrub species were found (oak individuals, being part of the understory layer, and 2 chamaephytes, because of their lignified growth habit, were included; see the complete list in Table A.1). Shrub species richness increased linearly along the chronosequence ($F_{1,11} = 56.91$, $P < 0.0001$; Fig. A.1A) from an average of 6.25 ± 3.00 species per plot in grazed plots to 15.67 ± 1.15 species in the 13-years excluded plots. Similarly, species diversity increased linearly with time since exclusion ($F_{1,11} = 27.59$; $P = 0.0003$; Fig. 2A), increasing from 3.44 ± 1.82 effective species to 9.23 ± 1.68 in the 13-years excluded plots. Across the whole chronosequence, *Ulex australis* (a spiny legume) and *Cistus salviifolius* (a dry-fruited species) were dominant in most sites, both grazed and excluded. Nevertheless, with increasing time since exclusion, other species from the *Cistaceae* family such as *Cistus crispus* and *Halimium halimifolium,* as well as some fleshy-fruited species such as *Myrtus communis,* became common. Species evenness varied between 0.56 and 0.59, and showed no tendency along the chronosequence ($F_{1,11}$ = 0.04 P = 0.840, data not shown).

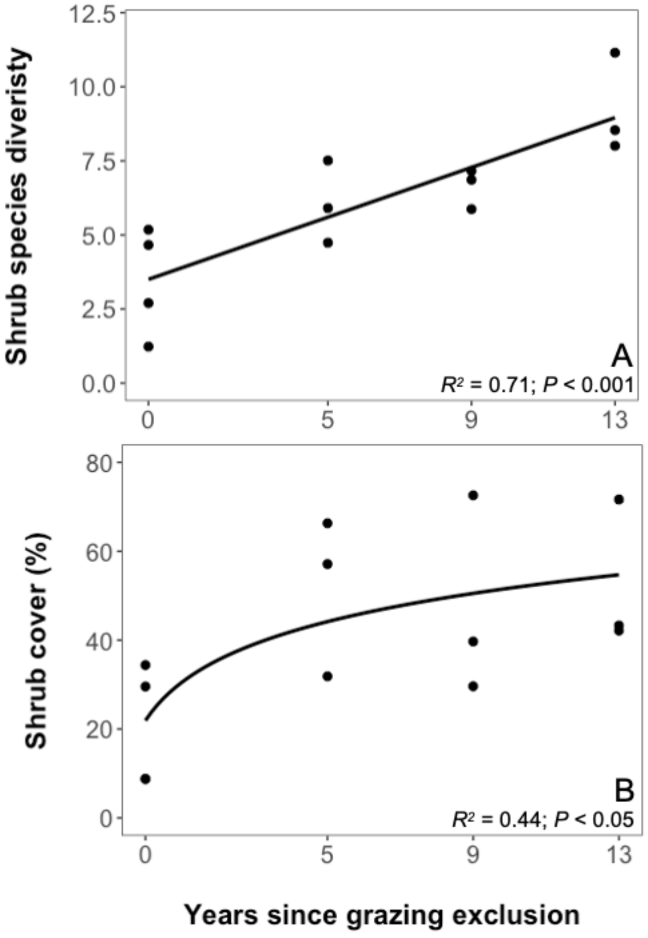

Figure 3: Shrub diversity and structure, measured by (A) species diversity, D^1, increasing linearly along the chronosequence, and (B) shrub cover, increasing after exclusion with a logarithmic function. Fitted lines are the LM, N = 13, at α = 0.05.

Shrub cover increased with exclusion following a logarithmic function ($F_{1,11}$ = 8.57, P = 0.014; Fig. 2B), increasing from an average of 20.38% ± 13.54 in grazed plots to 51.76% ± 17.84 in the 5-year excluded plots, and remaining relatively stable thereafter. Shrub height was, on average, 62.13 ± 11.99 cm, and did not change significantly along the chronosequence ($F_{1,11}$ = 2.14, P = 0.17, Fig. A.1B).

3.3. Functional diversity

Shrub functional richness increased along the chronosequence, following a linear function ($F_{1,10}$ = 6.60, P = 0.028;
Figure 4A).

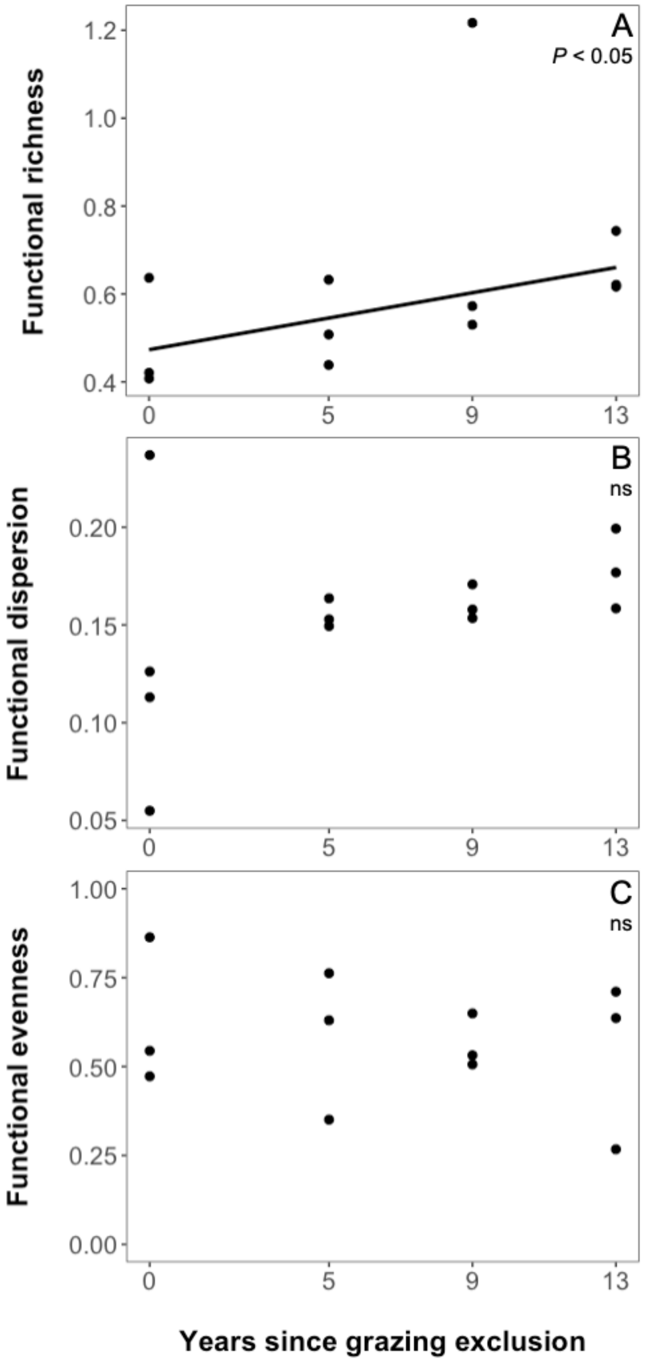

Figure 4: Shrub functional diversity indexes. (A) Functional richness, increasing significantly along the chronosequence; (B) functional evenness and (C) functional dispersion, both showing no pattern with age of exclusion. GLM, *N* = 12 for f. richness and f. evenness (computation is not possible for one grazed plot composed of only one species), *N* = 13 for f. dispersion, at α = 0.05.

Functional dispersion showed a marginally significant increasing trend with time since exclusion, and functional evenness did not change significantly ($F_{1,11}$ = 3.80, P = 0.077, and $F_{1,10}$ = 0.36, P = 0.560, respectively; Fig. 5B, 5C). Concerning individual trait CWM and f. dispersion, no significant changes were found with age of exclusion (Table A.2).

4. Discussion

Tree regeneration

Our results show that, in the studied cork oak woodlands, oak regeneration was hindered in grazed plots and in the 13-years excluded plots. Although not limited by seelings germination, as suggested by the presence of seedlings and saplings throughout the chronosequence, but limited by a lack of growth of saplings into young trees. A successful tree regeneration involves germination of acorns, establishment of seedlings and their growth into young trees (Ramírez and Díaz, 2008). In this work, our aim was to access tree regeneration from seedlings to new adult trees, by measuring the abundance and height of seedlings and saplings. In grazed areas, cattle may impose direct disturbance through soil compaction and browsing (López-Sánchez et al., 2016). Grazing exclusion, on the other hand, particularly in the 9-yrs excluded plots, showed the presence of taller saplings. Likely, seedlings germinated during grazing, or at an early exclusion period, were able to grow once grazing has ceased. However, this positive effect on tree regeneration was not observed on the 13-years excluded plots, where the height of saplings decreased and was similar to grazed plots. These findings suggest that other factors, co-varying with time since grazing exclusion, may be also influencing the growth of oak saplings. Shrub cover and diversity, which changed due to grazing exclusion, can influence, either positively or negatively, the regeneration of oak trees (Dias et al., 2016). Shrub cover increased from grazed to excluded plots, remaining then stable over time after exclusion. This may have had a positive influence on oak regeneration in the first 9 years of exclusion, but a negative influence thereafter. Shrubs can provide shade and shelter to seedlings, protecting them from browsing and, maybe even more importantly, summer drought (Arosa et al., 2015; Simões et al., 2016), although some species may have a stronger facilitative effect than others (Rolo et al., 2013). An amelioration of drought effects may also be provided by biological soil crusts, which contribute to soil water retention and temperature regulation in these systems, and are negatively affected by grazing (Concostrina-Zubiri et al., 2017).

Yet, depending on the circumstances, shrubs may also negatively impact oak regeneration by providing habitat for acorn predators and promote seedling mortality through competition, particularly in low diverse and dense shrublands (Acácio and Holmgren, 2014). Competition from shrubs could provide an explanation for the decrease of oak saplings' height from 9 to 13-years excluded plots. Listopad et al. (2018), who conducted a precedent study in the same plots, showed that vertical structural diversity of the plant community increased along our chronosequence. Thus, although shrub cover was similar among exclusion periods, the diversity of heights across the vertical profile of the system increases with increasing period of exclusion, driven by shrubs and also by the young cork oak trees that had grown during the last 13 years. In accordance with our results, Santana et al. (2011) showed these systems develop a highly diverse vertical structure within the first 15 years after disturbance cessation. This may promote competition with oak saplings for space, light, nutrition or water, limiting the sapling’s growth, and thus, limiting a continuous rate of tree recruitment along time. Other works in oak woodlands have shown a similar non-constant pattern of regeneration. Pons & Pausas (2006) suggested that oak regeneration, after *montado* abandonment, occurs in pulses, or temporal windows of opportunity - saplings develop into young trees within limited periods of time, when conditions are favourable, provided that a constant seedling bank is present. Oak regeneration requires conditions that are likely to change along succession. For example, certain shrub species facilitate oak seedlings, while others have neutral or negative effects, and these effects may acute as saplings grow (Rolo et al., 2013). Dias et al. (2016) noted that facilitation effects of certain shrub species may turn into competition at higher shrub cover values. This could explain a non-constant oak recruitment rate over time. Further studies, addressing directly the effects of shrubs on oak regeneration could evaluate a possible competing interaction over time.

Several studies on holm oak dominated *montados*, a similar ecosystem, have advocated temporal set-aside periods of 20-30 years to restore a consistent tree regeneration and a tree population structure with balanced numbers of young and old trees (Plieninger et al., 2003; Ramírez and Díaz, 2008). In our study, regeneration was improved after 9 years of exclusion, but it decreased in the 13 years excluded plots. A longer time series could confirm if regeneration would resume in a more consistent manner with time. Nevertheless, our results point to the usefulness of a temporal grazing exclusion of about 5-10 years to promote the growth of new cork oak trees. Similarly, other studies have found positive effects on cork oak regeneration within periods of less than a decade of no disturbance (or very low disturbance) (Dias et al., 2016; Simões et al., 2016). Additionally, our results suggest after 9 years of exclusion, the subsequent development of the shrub layer (driven by shrubs and by new young trees that have grown) may compete with oak saplings. At this point, it could be beneficial to install tree protectors around cork oak saplings and re-introduce grazers in the field. This would open vegetation, potentially promoting tree regeneration along time, while preventing the damage of oaks saplings by cattle (particularly by cows). Alternatively, localized shrub clearing**s** around oak saplings, could alleviate shrub competition, while maintaining shrub patches that protect seedlings at early stages.

The *montado* is a savanna-like system of sparse trees, with an understory of shrubs and pastures, traditionally managed in shifting rotations of grazing areas, annual crops and fellow lands. This promotes high spatial heterogeneity and maximizes the provision of services such as cork and cattle raising (Bugalho et al., 2011a; Ramírez and Díaz, 2008). In this context, a temporal grazing exclusion of 5-10 years, applied in areas where tree regeneration is insufficient, may be a passive ecological restoration measure, as a measure that assissts the recovery of ecosystem components, namely tree regeneration and a balanced maintenance of regeneration, shrub cover and pasture areas. Moreover, the growth of new cork oak trees is essential for the sustainability the *montado* system, from which the most important revenue is cork extraction. By promoting tree regeration, temporal grazing exclusion is also promoting montado’s market value to land-owners, contributing in this way to the conservation of this high nature value farmland (Bugalho et al., 2011a).

Our results should be taken as purely indicative when applied to other areas. Our work was conducted in a single study area. This allows to focus on processes occuring over time with good control of confounding factors, but limits extrapolation to other cork oak populations. Moreover, our study was conducted in permanent grazing excluded areas, thus we did not directly assess the effects of resuming grazing after exclusion. Some factors that may influence the effects of grazing exclusion on oak regeneration are, firstly, the presence of oak individuals and shrubs in grazed plots. In other areas regeneration may take longer if the area is completely deprived of seedlings and shrubs (Ramírez and Díaz, 2008). Secondly, climate and site history also influence the recovery patterns and tree growth rate: drier areas, shallow soils, and fire recurrency may also impact regeneration rates (Acácio and Holmgren, 2014; Príncipe et al., 2014). Although wild fires are not frequent in grazed montado areas, a higher cover of woody vegetation after grazing exclusion may increase fire recurrency, potentially forming persistent fire-prone shrublands that are hardly converted into oak woodlands (Acácio and Holmgren, 2014). Re-opening the field to grazers or alleviating shrub cover may also prevent the increase of fire risk associated with high fuel load (Santana et al., 2011).

Shrub diversity

Our results suggest that grazing exclusion allowed the secondary succession to preceed, with increasing shrub cover and diversity. Expectably, secondary succession of *montado* would lead the system to a Mediterranean oak forest or maquis (Acácio and Holmgren, 2014; Santana et al., 2011). In *montado* areas elsewhere, particularly in more arid areas and poor soils, abandonment has led to a mono-specific shrub encroachment – a highly resilient “arrested succession” state (Acácio and Holmgren, 2014). Nevertheless, in a context of grazed oak woodlands, and with the aim of improving plant diversity, our results suggest that excluding areas from grazing, particularly over a decade or more, may be effective for shrub diversity conservation. Other taxa, such as invertebrates and birds, are likely to benefit from shrub-rich excluded areas as well, and particularly from fleshy-fruited species that were common in the 9 and 13-years excluded plots (Bugalho et al., 2011b). Moreover, at the landscape scale, a matrix of grazed and excluded areas can further promote biodiversity by increasing spatial heterogeneity and landscape connectivity (Gonçalves et al., 2012).

In our study, shrub functional richness increased with increasing period of exclusion, matching the pattern of taxonomic diversity. The latter increases the functional space volume if the community has species with more extreme trait values (Mouchet et al., 2010). Functional dispersion, which considers species relative abundances,showed only a marginally significant increase with increasing time since exclusion, and f. evenness did not change along the chronosequence. This suggests that, although new species, and trait values, occur in the community with grazing exclusion (thus increasing functional richness), these new species are not much abundant, contributing little to alter shrub functional diversity in a consistent manner (i.e. increasing simultaneously richness, dispersion and evenness). However, the increasing trend of functional dispersion may suggest that as time since grazing passes, the abundance of species with different trait values increases, potentially augmenting functional dispersion at longer exclusion periods (Villéger et al., 2008). These results suggest that grazed plots preserve most of the functional diversity of 13-years excluded plots. This may be because there is no ecological space, or niches, to accommodate a higher functional diversity (Cadotte et al., 2011), or because a longer exclusion period would be needed to improve shrub functional diversity. Some species take longer to establish in a community, such as late successional species (Santana et al., 2011). The latter hypothesis is supported by our findings: species richness and functional richness increased linearly over 13 years of exclusion, indicating that the community has not stabilized yet. Mediterranean communities are typically slow growing, thus a longer monitoring period could further enlighten the relationship between time since grazing exclusion, taxonomic diversity and functional diversity.

Taken together, our results showed that grazed plots, compared with grazing excluded plots, had lower shrub cover and diversity. Plots excluded from grazing for 9 years had also taller oak saplings than grazed and 13-years excluded plots, suggesting that exclusion promoted oak regeneration in a period of 9 years, after which shrubs may hinder a continuous oak regeneration. Thus, promoting tree regeneration with the objective of cork exploration may be accomplished with this period of exclusion, particularly if there are no limitations of germination. Along 13 years of exclusion, shrub species diversity increased linearly, but no consistent changes were found concerning shrub functional diversity, except for functional richness that also increased with time since exclusion. In the context of the studied cork oak woodlands, temporal grazing exclusion can thus be an effective passive restoration measure, promoting the recovery of tree regeneration and shrub diversity. The diverse patterns observed for different vegetation components along time since exclusion, point to the importance of objective-oriented restoration measures. Restoration focused on the provisioning ecosystem services, such as oak regeneration, may require a shorter period of exclusion, while restoration of regulating and maintenance services, such as the maintenance of biodiversity, may require longer periods of grazing exclusion (i.e. a decade or more).

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