

Title: Conservation zones promote oak regeneration and shrub diversity in certified Mediterranean oak woodlands

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Abstract

Mediterranean oak woodlands are ecosystems of high conservation and socio-economic value that occur in Southwestern Europe, North Africa and California. Oak regeneration failure is occurring in these ecosystems and may be endangering their long-term conservation. Most studies suggest that inadequate management practices may be contributing to oak regeneration failure. Forest certification is a voluntary type of certification, based on third-party auditing of compliance with performance-based sustainable management standards that has been expanding in forest ecosystems worldwide, including in Mediterranean oak woodlands. The Forest Stewardship Council (FSC) certification is the dominant certification scheme in Mediterranean oak woodlands and requires landowners to establish conservation zones in their estates. Conservation zones usually correspond to a tenth of the estate and are primarily managed for biodiversity conservation. In spite of recent studies reporting positive effects of FSC certification and conservation zones on biodiversity and forest structure in tropical regions, its effects on tree regeneration in Mediterranean oak woodlands are unknown. In this study, conducted in Southwestern Europe, we compared the abundance of cork oak (*Quercus suber*) regeneration and the cover, richness and diversity of Mediterranean shrublands between conservation and non-conservation zones in FSC certified cork oak woodlands. We found that in conservation zones oak regeneration was more abundant and that species richness and diversity of shrubs were significantly higher. Our results suggest the creation of set-aside areas in cork oak woodlands, such as conservation zones, may help avert the tree regeneration crisis this ecosystem is facing.

Keywords: cork oak, distance sampling, Forest Stewardship Council, Mediterranean shrublands, *Quercus suber*, tree regeneration

1) Introduction

Mediterranean oak woodlands are integrated in the World Biodiversity Hotspots of the Mediterranean Basin and the California Floristic Province (Myers et al., 2000). They occur in the Mediterranean regions of California, Southwest Europe and North Africa (Huntsinger et al., 2013) harbouring significant concentrations of endemic and threatened species (Díaz et al., 2013). These ecosystems are characterized by a savanna type structure with a diverse understory of shrublands intermixed with grasslands with high heterogeneity (Miguel Nuno Bugalho et al., 2011; Díaz et al., 2013). These ecosystems are also socio-economically important, generating a variety of services such as livestock production, long-term carbon storage, hunting or recreation (Caparrós et al., 2013).

Oak regeneration failure has been reported in Mediterranean oak woodlands globally and associated with inadequate management practices (e.g. overgrazing and intensive shrub clearing to reduce risk of wildfires), which may be endangering the conservation of these ecosystems. For example, in California overgrazing and shrub encroachment, combined with the introduction of annual exotic grasses are negatively affecting oak regeneration (Tyler et al., 2006). In Southwest Europe, there are two main groups of factors causing regeneration failure: 1) localized overgrazing and excessive shrub clearing in some regions and 2) lack of management, which causes shrub encroachment and increases wildfire risk in others (Acácio et al., 2007;

Pulido et al., 2010). In North Africa overharvesting of wood, overgrazing and overcollection of acorns for human and animal consumption are causing oak regeneration failure (Campos et al., 2007).

Some authors suggest that setting aside areas with low or no management, initiating secondary succession processes where shrub cover increases, can protect and enhance oak regeneration (e.g. Ramírez and Díaz, 2008; Rey Benayas et al., 2008). Decreasing herbivory and trampling by cattle may promote oak regeneration (Plieninger et al., 2010) but may also increase shrub cover, where acorn predation by mice is high (Acácio et al., 2007). Also, there are species-specific effects that can affect oak regeneration. For instance, shrubs such as *Retama* spp. and *Ulex* spp. can ameliorate the effects of harsh temperatures and light conditions, improving seedling survival, while others such as *Cistus ladanifer* L. have been reported to compete with oak seedlings and decrease seedling survival (Acácio et al., 2007). In this context, it is important to understand the relative importance of competing or facilitating effects of shrubs on cork oak regeneration and to assess how such effects will balance with, for example, effects of predation by small rodents in these set-aside areas.

Forest certification is a voluntary conservation mechanism, based on third-party auditing of compliance with performance-based sustainable management standards (Auld et al., 2008). Under Forest certification landowners and managers have to comply with a set of sustainable management standards that include environmental and socio-economic criteria (Auld et al., 2008). Products generated in areas under forest certification are labelled for consumer recognition and have higher market

value (Gulbrandsen, 2010). Certification has been expanding worldwide with the two largest certification schemes, the Program for the Endorsement of Forest Certification (PEFC) and the Forest Stewardship Council (FSC), covering currently 251 and 186 million hectares, corresponding to 6.27 and 4.65% of the world's forests, respectively (FSC, 2014; PEFC, 2014).

The dominant certification scheme in Mediterranean oak woodlands is FSC certification, which requires landowners and managers to establish conservation zones in their estates. These zones usually correspond to a tenth of the estate area and are primarily managed for biodiversity conservation (Tollefson et al., 2009). Studies on the impacts of FSC certification and conservation zones are mostly limited to tropical regions (e.g. Bennett, 2001; Gullison, 2003) and generally conclude that FSC certification has positive effects on forest conservation (Putz et al 2012). There are very few studies on the impacts of FSC certification on temperate forests and Mediterranean oak woodlands. For instance, Dias et al. (2015) found that FSC certification benefited the ecological condition of Mediterranean streams crossing certified cork oak woodlands.

Cork oak woodlands are typical of the Western Mediterranean Basin and cover 1.5 million hectares in Southwest Europe and 1 million hectares in North Africa (J. Pausas et al., 2009). These ecosystems harbour high levels of biodiversity (Díaz et al., 1997) and provide cork, a non-timber forest product harvested every 9 to 12 years which is mainly used for bottle stoppers (over 70% of the production) but also in other uses such as insulation, flooring or decoration (Aronson et al., 2009). As in other Mediterranean oak woodlands, low cork oak regeneration is threatening this

ecosystem (Pausas et al., 2009). The largest cover by cork oak occurs in Portugal (approximately 736 000 ha corresponding to 40% of its distribution) where c. 100 000 ha are under FSC certification. In FSC certified conservation zones landowners are allowed to harvest cork but livestock grazing and shrub clearing, for example, are halted or significantly reduced. Outside conservation zones these activities are allowed, but must be conducted according to environmental standards that include, for example, not using heavy machinery (e.g. disc harrows) when clearing shrubs.

The main purpose of the present study was to assess whether the creation of conservation zones in FSC certified areas in Mediterranean oak woodlands can promote oak regeneration. We compared 1) the abundance of seedlings, saplings and young cork oak trees and 2) the cover and diversity of Mediterranean shrublands, in conservation zones and adjacent non-conservation zones in FSC certified cork oak woodlands.

2. Methods

Study area

The study area is located in Southern Portugal, in the Alentejo region. The terrain is relatively hilly with a mean altitude of 54 metres above sea level. The climate is sub-humid Mediterranean, with a mean annual temperature of 16°C and an average rainfall of 730 mm/year (AEM and IM, 2011). The dominant soil types are litholic humic and non-humic soils and podzols (Cardoso, 1974).

Cork oak woodlands are the dominant forest type in this region (42% of the study area) (Fig. 1) and are characterized by a sparse tree cover of cork oak, mixed with holm oaks *Quercus ilex rotundifolia* Lam. or pine trees and a diverse understory of shrublands (e.g. rockroses *Cistus* spp., gorse *Ulex* spp., basil-leaved rock rose *Hallimium ocymoides* Willk., topped lavender *Lavandula stoechas* L. and rosemary *Rosmarinus officinalis* L.) interspersed with grasslands (e.g. *Agrostis* spp., *Avena* spp., *Bromus* spp.), pastures, fallows and cereal crops (Bugalho et al., 2009). Other landuses include agricultural areas and pastures (27%), blue gum *Eucalyptus globulus* Labill plantations (9%) and stone pine (*Pinus pinea* L.) and maritime pine (*Pinus pinaster* L.) plantations (12%).

Field sampling

We surveyed eight FSC certified cork oak woodland estates, seven of which with six years of certification and one with four years (Table 1). Across these estates we surveyed a total of 14 conservation zones, (Fig. 1) for oak regeneration, shrub cover, richness and diversity and adult tree cover, using line transects, as described below. We also surveyed areas adjacent to conservation zones with approximately same size and similar shape, henceforth referred to as non-conservation zones. Three to five 50 metre line transects were randomly established in each conservation and non-conservation zone. Overall we established 86 transects, 41 in conservation zones and 45 in non-conservation zones (Table 1). To estimate the abundance of cork oak regeneration we used distance sampling (Buckland et al., 2001). Distance sampling is a method for estimating population density based on surveying a number of randomly located transects. An observer

177 travels along these transects and measures the perpendicular distance between
178 each observation and the line. The method assumes the detection probability of an
179 individual decreases with distance from the line, objects are stationary, all individuals
180 located on the line are detected and distances are measured without error (Thomas
181 et al., 2010). The distribution of the distances is used to estimate a detection
182 function, expressing the probability of detecting an individual at a given distance.
183 Numerical and factor covariates (e.g. plant height or habitat type) can be included in
184 the detection function to improve model fit (Marques et al., 2007). The probability of
185 detection can be incorporated into a robust (Horvitz-Thompson-like) density
186 estimator of abundance (Thomas et al., 2010). We recorded cork oak seedlings,
187 saplings and young trees as well as the height of each recorded plant. Plants were
188 classified as seedlings ($\text{height} \leq 10 \text{ cm}$), saplings ($10 \text{ cm} < \text{height} \leq 50 \text{ cm}$) and
189 young trees ($50 \text{ cm} < \text{height} \leq 4 \text{ m}$ and diameter at breast height $\leq 20 \text{ cm}$) following
190 the classification proposed by Pons and Pausas (2006). We used this classification
191 since it was developed for a region in Spain whose edaphoclimatic characteristics
192 are similar to those in our study area, where cork oak growth is likely to be similar.
193 We are aware that distance sampling is not common in vegetation surveys, despite
194 the fact its assumptions are easily met in these studied, as there are no evasive
195 movements and distances can be measured accurately (Buckland et al 2001).
196 Distance sampling can offer some advantages relatively to conventional methods
197 such as quadrat sampling, which consists in counting all individuals in a given
198 sampling unit that can be of rectangular or circular shape (Krebs, 1998). Distance
199 sampling does not assume that all individuals are counted, regardless of habitat
200 conditions, which may not be possible for small plants and incorporates this
201 uncertainty in the estimates. Also, surveyors can prospect larger areas in shorter

periods, which is particularly important in highly heterogeneous areas such as Mediterranean oak woodlands.

We used the line intercept method to measure (relative) shrub cover (Canfield, 1941; Krebs, 1998). This method consists in recording shrub species intercepted by the transect and recording lengths of shrub canopy projections along the transect. Shrub cover is calculated by dividing the intercept lengths of a species by the length of the transect. The diversity of shrub species was calculated using the Shannon-Weaver diversity index (Krebs, 1998).

We used the point-centered quarter method to measure (absolute) tree cover (Pollard, 1971). This method requires selecting a number of points along the transect and dividing the area around each point in four 90° quadrants. The distance between each points to the nearest tree, in each quadrant, is measured. We selected one point at the beginning, one at the middle (25 metres) and one at the end of each transect to avoid measuring the same individual at two successive points. Only trees with a diameter at breast height ≥ 20 cm (adult trees) were considered. The diameter at breast height of each selected tree was recorded to calculate the basal area of the trees, which is a proxy for tree cover. With this information, using the point-centered quarter method, we computed the absolute cover of each tree species. Cork had the highest relative tree cover ($> 90\%$) both on conservation and non-conservation zones and the relative cover by other species was negligible. All field sampling work was conducted by the same 2 observers (FSD and JM).

Density surface models

227 To model the effect of conservation zones on the abundance of cork oak seedlings,
228 saplings and young trees we used density surface modelling (DSM) (Hedley and
229 Buckland, 2004; Miller et al., 2013). This is a two-stage approach that involves 1)
230 fitting a detection function to the distance data and using it to estimate the
231 abundance of oak regeneration in each transect and 2) building a generalized
232 additive model (Wood, 2006) to relate oak abundance per transect to environmental
233 covariates.

234

235 Uniform, half-normal and hazard-rate detection functions were fitted to the data.
236 Along with the adjustment terms, height of oak plants, shrub cover, tree cover were
237 included as covariates (one or two variables at a time) as they may affect plant
238 detectability. “Zone code”, a 27 level categorical variable with levels corresponding to
239 each surveyed conservation and non-conservation zone, was also added as a
240 covariate. Observed distances were truncated at six metres based on visual
241 inspection of the detection function superimposed on the histogram of distances
242 (Buckland et al., 2001) (Appendix A). The best detection function was selected using
243 Akaike’s Information Criteria (AIC). All analyses were implemented using the
244 Distance package version 0.9 for R version 3.0.1 (R Core Team, 2014). Oak
245 regeneration abundances were estimated per transect from the detection function
246 using a Horvitz-Thompson-like estimator (Borchers et al., 1998).

247 Models for the abundance of seedlings, saplings and young trees per transect were
248 implemented using Generalized Additive Models (GAM) (Wood, 2006). Expected
249 abundances in each transect were assumed to follow a Tweedie or quasi-Poisson
250 distribution. The Tweedie power parameter was estimated during model fitting. The
251 explanatory variables included the variable “zone type” which has two levels

corresponding to conservation and non-conservation zones. Variables were divided in two groups “local” and “topographic”. This division allowed us to analyse the effects of local and topographic variables on cork oak regeneration, while minimizing correlation problems.

Local variables included absolute cover of adult trees and the cover, species richness and diversity of shrubs. For assessing effects of shrub cover on cork oak regeneration we only considered the most frequent and dominant shrubs (i.e. occurring in more than 20% of transects) as those are the most likely to affect positively (facilitation) or negatively (competition) cork oak regeneration. Species richness and diversity were calculated using all recorded species. Shrub species that occurred in more than 20% of transects were: *Cistus salviifolius* L., *C. crispus* L., *Ulex spp.*, *Rosmarinus officinalis* L., *C. monspeliensis* L. and *Lavandula stoechas* L.. We used individual species shrub cover and not total shrub cover in this analysis as shrubs may have facilitative or competitive effects on cork oak regeneration according to species identity. Species richness and species diversity were calculated using all encountered species. The topographic variables, which include slope, aspect and soil type were gathered from a Digital Elevation Model (METI and NASA, 2011) and Portugal's Soil Chart (Cardoso, 1974) (Table 2) using QGIS 2.2 (QGIS Development Team, 2014). Each group of variables was used to fit the local and topographic variables.

GAMs were fitted with the *dsm* package (Miller, 2014) for R 3.0.1 (R Core Team, 2014). Thin plate regression splines were used as the basis for the model's smooth terms. Basis complexity was selected by specifying an overly wiggly basis and then

letting the penalty select the correct wigglyness of the term (Wood, 2006). To assess the degree of correlation and multiple correlation between the covariates, we calculated variance inflation factors (VIF) (Fox and Weisberg, 2010), all covariates had a VIF <3. Smoothness selection for the smooth terms was performed via restricted maximum likelihood (REML), because the REML criteria tends to have a more pronounced optima (Wood, 2011). The area surveyed in each transect multiplied by the average detection probability was used as an offset to account for effort expended. An estate identifier was included as an eight level random effect to account for non-independence in abundance within the same estate (Wood, 2013). Smooth terms were selected using approximate p-values ($p < 0.05$) and an extra penalty was included in the model that allowed each smooth term to be removed during model fitting (Marra and Wood, 2011). Deviance residuals were checked for normal distribution and constant variance (Wood, 2006). Spatial autocorrelation was assessed by examining a correlogram built with model residuals using function “correlogram” from the R package “spatial” (Venables, 2003).

Shrub cover, richness and and diversity

For comparing the cover and diversity of shrub species between conservation and non-conservation zones, we applied linear mixed effects models using the R package “nlme” (Pinheiro and Bates, 2009). Shrub cover, richness and diversity were used as response variables and “zone type” (categorical variable) as the independent variable. We checked the model residuals for violations of normality and homogeneity and for spatial autocorrelation (Zuur et al., 2009). When homogeneity was violated a

“VarIdent” variance structure was added to the model. To check for spatial autocorrelation a semivariogram was analysed. When spatial autocorrelation was detected we added a Gaussian correlation structure to the model. We determined if the random effects of the models were normally distributed by histogram inspection (Pinheiro and Bates, 2009).

3. Results

We counted 2409 cork oak plants in the 86 line transects. From these, 717 were seedlings, 1028 were saplings and 332 were young trees. The density of seedlings, saplings and young trees in conservation and non-conservation zones was (mean \pm standard error of the mean) 1154 ± 144 plants ha^{-1} and 850 ± 173 plants ha^{-1} , 1639 ± 259 and 770 ± 119 plants ha^{-1} and 170 ± 33 plants ha^{-1} and 126 ± 31 plants ha^{-1} , respectively.

Detection function

The hazard rate detection function with plant height and zone code as covariates was selected by AIC. The other functions were poorer ($\Delta\text{AIC} > 59$; Appendix A, Table A.1). The truncation distance for the detection function was 6 metres and chosen by analysing how abundance estimates changed with the truncation distance and by comparing test statistics from the Cramer-von Mises and Kolmogorov-Smirnov goodness of fit tests (Buckland et al., 2001) (Appendix B). The average detection probability was 0.338 and its coefficient of variation was 0.03. A more detailed description of the selection process of candidate detection functions can be found on

Appendix B, where part of this dataset was used to exemplify the use of distance sampling methods in plant surveys.

Local density surface models

DSMs with Tweedie distributions were selected. Seedling abundance was significantly higher in conservation zones and increased until cork oak cover reached 0.0025 m² ha⁻¹, lowering afterwards (Figure 2). Sapling abundance was significantly higher in conservation zones, increasing until the cover of shrub *Ulex* spp. Reached 11.5% decreasing afterwards (Figure 3), and was negatively affected by *R. officinalis* cover (Figure 3). The abundance of young trees was significantly higher in areas of higher shrub diversity (Table 3 and Figure 4)

Topographic density surface models

DSMs with Tweedie distributions were selected for seedlings and saplings. For young trees a quasi-Poisson distribution was selected. No topographic variables had an effect on the abundance of seedlings. The abundance of saplings was higher on the western part of the study area (Figure 5). The abundance of young trees was significantly higher in areas with azimuth values of ~ 220 degrees (Figure 6), that is, in areas with a Southwest orientation. The abundance of young trees was also significantly lower in areas with higher percentages of cover of litholic soils and podzol soils (Figure 6 and Table 3).

Comparing shrub cover and diversity between conservation and non-conservation zones

There were no significant differences in shrub cover between conservation (mean \pm standard error of the mean) (12.46 ± 2.04 %) and non-conservation zones (10.05 ± 2.55 %). However, species richness (3.64 ± 0.28 vs 2.66 ± 0.65) and diversity (0.935 ± 0.10 vs 0.667 ± 0.13) of shrubs were significantly higher in conservation zones as compared to non-conservation zones. As for individual species, the cover of *Ulex* spp. was significantly higher in conservation zones (3.467 ± 0.273 vs 1.386 ± 0.100), but no significant differences were found for other species (Table 4).

4. Discussion

Our results suggest the implementation of conservation zones on certified cork oak woodlands had a positive effect on oak regeneration and shrub richness and diversity. Conservation zones are set-aside areas with less intensive management where livestock grazing and shrub clearing is halted or reduced. Grazing is important for maintaining the open structure of woodlands and promotes habitat heterogeneity (Miguel N. Bugalho et al., 2011; Veldman et al., 2015), but high grazing pressure, decreases seedling survival and reduces oak regeneration (Plieninger, 2007). Over long periods, livestock grazing may reduce soil organic matter and cause soil compaction (Belsky et al., 1999), making it harder for young cork oak roots to grow and obtain water (Serrasolses et al., 2009). Shrub clearing is conducted in cork oak woodlands to reduce the risk of wildfires (Bugalho et al., 2009). However, it should be conducted over long rotation periods (e.g. 4 to 7 years) to allow for oak

regeneration and recruitment (Aronson et al., 2009). Higher abundance of seedlings and saplings in conservation zones may be associated with low or no livestock grazing and less frequent shrub clearing. The period during which the surveyed estates were under forest certification (maximum of 6 years), however, did not allow for significant differences to be detected in the abundance of young trees. This is unsurprising as cork oaks are slow growing trees (Pausas et al., 2009). It may also suggest that conservation zones were not created in areas of particularly high cork oak regeneration.

Both grazing and shrub clearing reduction may explain the higher species richness and diversity of shrubs in conservation zones. It has been shown that long-term high grazing pressure reduces the species richness of the seed bank and the species richness of the above ground vegetation (e.g. Chaideftou et al., 2009). Shrub cover was similar in conservation and non-conservation areas, which may suggest grazing pressure differences and/or certification time may not have been enough to induce significant differences. The only shrub species whose cover increased in conservation zones were *Ulex* spp., which are very palatable species for livestock during its early stages of development when its spines have not hardened (Rodwell, 1998).

The abundance of seedlings increased with higher cork oak cover, which may be explained by a higher concentration of acorns beneath the tree canopy (Weltzin and McPherson, 1999). Positive effects on the microclimate under tree canopy may have also occurred. Drought is a key factor determining seedling survival in Mediterranean climates (e.g. (Gómez-Aparicio et al., 2005)). Tree canopy may facilitate seedling

401 survival by protecting seedlings from high temperatures (Caldeira et al., 2014;
402 Puerta-Piñero et al., 2007) thus decreasing the need of water to transpire. Other
403 indirect interactions include higher water availability through hydraulic lift (Brooker et
404 al. 2008) or decreased competition with herbs whose biomass tend to be lower
405 under oak canopies (Caldeira et al., 2014). Interestingly, seedling abundance did not
406 respond to increased tree cover after a threshold of $0.0025 \text{ m}^2\text{ha}^{-1}$ was reached. This
407 may indicate that at such a tree cover competitive interactions between adult oak
408 trees and seedlings may be prevalent (Plieninger et al., 2010).

409

410 Sapling abundance was higher in areas with intermediate levels of *Ulex* spp. cover
411 but decreased with increasing cover by *Rosmarinus officinalis*. *Ulex* spp. are spiny,
412 perennial, evergreen shrubs that can fix nitrogen and provide physical protection
413 against livestock and shade to seedlings (Gómez-Aparicio et al. 2004). *R. officinalis*
414 is an obligate seeder which is very competitive for water and soil nutrients, due to its
415 high ratio of root length to total plant biomass (Hernández et al., 2010). These traits
416 may explain the negative effect of *R. officinalis* cover on the abundance of saplings.

417

418 The abundance of young trees seems to be positively associated with higher shrub
419 diversity. Similar results were found for other oaks in the Mediterranean region
420 (Plieninger et al., 2011). It has been shown that areas with a history of agriculture
421 combined with intensive grazing in Mediterranean regions tend to present low shrub
422 diversity due to seed bank depletion (Chaideftou et al. 2009). Some of these areas
423 may also show low levels of oak regeneration due to habitat degradation (Navarro-
424 González et al., 2013). Therefore, this result may suggest there is a higher
425 abundance of young trees in areas where grazing and agriculture have been

historically less intensive. This result may also indicate that shrub diversity increases in areas with a higher abundance of young cork oaks, possibly because of lower competitive interactions.

We did not find meaningful relations between the abundance of cork oak seedlings and saplings and topographic variables. Seedling survival is highly dependent on micro-environmental conditions (Plieninger et al., 2010) and probably the large scale nature of the topographic variables we used in this study does not translate into the finer regeneration niche scale of seedlings and saplings. The abundance of young trees was higher in areas with a Southwest orientation. Cork oak growth is particularly sensitive to water but also to light availability and older trees need enough light to be able to photosynthesise and produce photo-assimilates that exceed their respiratory needs (Caldeira et al., 2014). Also, cork oaks are particularly sensitive to frost and areas with a Southwest orientation are warmer and frost is less frequent (Pausas et al., 2009). Young trees are also less abundant in areas with higher cover by litholic non humic soils and podzols, which may be due to the fact these soil types have low organic matter content and are chemically poor (Serrasoltes et al., 2009) and therefore less suitable for cork oaks. Cork oaks also develop better in soils not compacted or flooded with a structure that permits good aeration (Serrasoltes et al., 2009).

Due to different sizes of conservation zones it was not possible to fully account for potential edge and dispersal effects and results could have been affected by this. Nevertheless, the overall results mirrored closely the results obtained for each estate, i.e., the abundance of seedlings and saplings, as well as shrub richness and

diversity were higher on conservation zones across the estates. Only a few exceptions were observed, as there were more seedlings on the non-conservation zones of the Arrão and Caniceira estates. In the same estates, shrub richness and diversity were also higher on the non-conservation zone.

The results of this study suggest the creation of conservation zones may help avert the tree regeneration crisis cork oak woodlands are facing. Increasing the area allocated to conservation zones could promote cork oak regeneration in cork oak landscapes but this would imply reducing livestock grazing areas, which could have significant negative economic effects. In such cases, compensations to landowners such as payments for ecosystem services (e.g. Bugalho et al., 2015; Wendland et al., 2010) could help balance the loss of financial returns from livestock grazing. Alternatively, rotating the location of conservation zones over periods of time allowing juvenile oaks to be established could be an alternative. Ramírez et al. (2008) showed that in Central Spain setting aside areas in cork oak and holm oak woodlands for periods of 20-25 years promoted oak regeneration and balanced numbers of old and young trees. Given that cork oaks have high longevity (>200 years), this could be a viable option for promoting cork oak regeneration.

Conclusions

We show that conservation zones in certified Mediterranean oak woodlands promote oak regeneration and understory diversity and that increasing the area allocated to conservation zones may contribute to increase oak regeneration at the landscape scale. This suggests that conservation zones can play an important role in ensuring

the long-term persistence of cork oak woodlands and promote their characteristic habitat heterogeneity on which several endemic and threatened species depend (Berrahmouni et al., 2009). We point out that these findings are likely to be highly conservative as we surveyed conservation zones in certified areas where management (even outside conservation zones) is already conducted according to environmental standards, which include the prohibition of using heavy machinery to conduct shrub clearings, even outside conservation zones. FSC certification is expanding rapidly in forest ecosystems, therefore assessing how conservation zones are contributing to forest conservation is crucial to inform the present and future implementation of certification standards.

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756

757 Table 1 – Number and area of the surveyed conservation zones (CZ) and non-
 758 conservation zones (NCZ)

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Estate	Years of certification	# CZ	Mean area of CZ (ha)	# Transects on CZ	# NCZ	Mean area of NCZ (ha)	# Transects on NCZ
Arrão de	6	1	44.78	3	1	44.78	3
Baixo							
Caniceira	6	1	19.81	3	1	19.81	3
Cascavel	6	1	20.64	3	1	20.64	3
Cavaleiros	4	2	19.48	6	1	19.48	3
Fidalgos	6	5	7.72	15	5	7.72	15
Machoqueira	6	2	100.54	9	2	100.54	8
do Grou							
Onzenas de	6	1	5.60	3	1	5.60	3
cima							
Pereira	6	1	8.49	3	1	8.49	3
TOTAL		14		45	13		41

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770 Table 2 – List of the variables included in the local and topographic density surface
771 models

Variables	Name	Description and units
Seedlings abundance		Plants/transect
Saplings abundance		Plants/transect
Young trees		
abundance		Plants/transect
	Zone type	Categorical variable with two levels, that distinguishes between “conservation zones” and “non-conservation zones”.
	Absolute cover of adult	
	cork oaks	m ² /ha
	Shrub richness	Number of species
	Shrub diversity	Shannon-Weaver diversity index
Local variables	<i>Lavandula stoechas</i> cover	% of cover
	<i>Rosmarinus officinalis</i>	
	cover	% of cover
	<i>Cistus salvifolius</i> cover	% of cover
	<i>Ulex</i> spp. cover	% of cover by <i>Ulex minor</i> and <i>Ulex australis</i>
	<i>Cistus crispus</i> cover	% of cover
	<i>Cistus monspeliensis</i> cover	% of cover
	Altitude	Metres
	Slope	Percentage
	Aspect	Azimuth
	Topographic ruggedness	
	index	
Topographic variables	Topographic position index	
	Latitude	
	Longitude	
	Litholic soils	Percentage of cover
	Clayey soils with low	
	saturation	Percentage of cover
	Podzol soils	Percentage of cover

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773 Table 3 – Density surface models built using local and topographic variables,
 774 selected covariates and deviance explained. Smooth functions denoted by s(). Zone
 775 type: conservation zone (+) denotes the abundance of seedlings, saplings or young
 776 trees was significantly higher on conservation zones.

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Scale	DSM	Distribution	Covariates	Deviance explained
Local	Seedlings	Tweedie	Zone type:conservation zone (+), Cork oak cover, s(Estate)	35.7%
	Saplings	Tweedie	Zone type:conservation zone (+), <i>Ulex sp</i> cover, <i>R. officinalis</i> cover, s(Estate)	59.4%
	Young trees	Tweedie	s(Shrub diversity), s(Estate)	48.9%
	Seedlings	Tweedie	Zone type:conservation zone (+), s(Estate)	24.4%
	Saplings	Tweedie	Zone type:conservation zone (+), s(x and y coordinates)	44.7%
Topographic	Young trees	Poisson	s(Aspect), s(Area of litholitic non humic soil), s(Area of podzol soils), s(Estate)	63.1%

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784 **Table 4** – Results from linear mixed effects models. The last column indicates the
785 type (if any) of the added spatial correlation structure. *P*-values < 0.05 are printed in
786 bold. “+” mean the metric had significantly higher values on conservation zones and
787 “-” the opposite. Estimated regression parameters, standard errors and t-values can
788 be found on Appendix B.

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Variable		p-value	Spatial correlation structure
Species richness		0.004 (+)	Gaussian
Shannon-Weaver diversity index		0.008 (+)	Gaussian
Shrub cover	Overall	0.448	
	<i>Rosmarinus officinalis</i>	0.790	
	<i>Cistus salvifolius</i>	0.318	
	<i>Ulex sp</i>	0.010 (+)	Gaussian
	<i>Cistus crispus</i>	0.919	
	<i>Lavandula stoechas</i>	0.645	
	<i>Cistus monspelliensis</i>	0.161	
	<i>Hallimium lasianthum</i>	0.299	

<i>Cistus ladanifer</i>	0.101
<i>Daphne gnidium</i>	0.628
<i>Quercus lusitanica</i>	0.455
<i>Pterospartum tridentatum</i>	0.180
<i>Rubia peregrina</i>	0.125
<i>Hallimium ocymoides</i>	0.317
<i>Erica australis</i>	0.363
<i>Pistacia lentiscus</i>	0.298
<i>Myrtus communis</i>	0.312

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Figure 1 – Location of the estates and of the surveyed conservation zones (purple) and non-conservation zones (grey). Land use inside and outside of the surveyed estates is also shown. All estates have Forest Stewardship Council Certification (FSC). Smaller pairs of conservation and non-conservation zones are highlighted with a circle. Note that on inset “2” two estates are represented, “Machoqueira do Grou” and “Arrão”.

823 Figure 2 - Local seedlings model - smooth function of the variable Cork oak cover
824 (m^2/ha).

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848 Figure 3 – Local saplings model - smooth functions for the variables *Ulex* sp cover
849 (%) and *Rosmarinus offinalis* cover (%) included in the local saplings model.

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872 Figure 4 – Local young trees model - smooth functions for the variable shrub
873 diversity.

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Figure 5 – Topographic saplings models – this image shows how the abundance of saplings changes with the x,y coordinates. Saplings abundance is shown in a colour gradient ranging from light grey (lower) to dark grey (higher).

921 Figure 6 – Topographic young trees model - smooth functions aspect (expressed as
922 azimuth value), cover of litholic soils (%) and cover of podzol soils (%).