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 South-western 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 16ºC 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 travels along these transects and measures the perpendicular distance between each observation and the line. The method assumes the detection probability of an individual decreases with distance from the line, objects are stationary, all individuals located on the line are detected and distances are measured without error (Thomas et al., 2010). The distribution of the distances is used to estimate a detection function, expressing the probability of detecting an individual at a given distance. Numerical and factor covariates (e.g. plant height or habitat type) can be included in the detection function to improve model fit (Marques et al., 2007). The probability of detection can be incorporated into a robust (Horvtiz-Thompson-like) density estimator of abundance (Thomas et al., 2010). We recorded cork oak seedlings, saplings and young trees as well as the height of each recorded plant. Plants were classified as seedlings (height ≤ 10 cm), saplings (10 cm < height ≤ 50 cm) and young trees (50 cm< height ≤ 4 m and diameter at breast height ≤ 20 cm) following the classification proposed by Pons and Pausas (2006). We used this classification since it was developed for a region in Spain whose edaphoclimatic characteristics are similar to those in our study area, where cork oak growth is likely to be similar. We are aware that distance sampling is not common in vegetation surveys, despite the fact its assumptions are easily met in these studied, as there are no evasive movements and distances can be measured accurately (Buckland et al 2001). Distance sampling can offer some advantages relatively to conventional methods such as quadrat sampling, which consists in counting all individuals in a given sampling unit that can be of rectangular or circular shape (Krebs, 1998). Distance sampling does not assume that all individuals are counted, regardless of habitat conditions, which may not be possible for small plants and incorporates this 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 90º 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

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

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

Models for the abundance of seedlings, saplings and young trees per transect were implemented using Generalized Additive Models (GAM) (Wood, 2006). Expected abundances in each transect were assumed to follow a Tweedie or quasi-Poisson distribution. The Tweedie power parameter was estimated during model fitting. The 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 ± 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 (ΔAIC > 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 Kolmogrov-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 m^2 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 ± 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 survival by protecting seedlings from high temperatures (Caldeira et al., 2014; Puerta-Piñero et al., 2007) thus decreasing the need of water to transpire. Other indirect interactions include higher water availability through hydraulic lift (Brooker et al. 2008) or decreased competition with herbs whose biomass tend to be lower under oak canopies (Caldeira et al., 2014). Interestingly, seedling abundance did not respond to increased tree cover after a threshold of 0.0025 m^2ha^{-1} was reached. This may indicate that at such a tree cover competitive interactions between adult oak trees and seedlings may be prevalent (Plieninger et al., 2010).

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

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

(Serrasolses 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 (Serrasolses 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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7. References

Acácio, V., Holmgren, M., Jansen, P.A., Schrotter, O., 2007. Multiple recruitment limitation causes arrested succession in Mediterranean cork oak systems. Ecosystems 10, 1220–1230.

AEM, IM, 2011. Iberian Climate Atlas. Agencia Estatal de Meteorología de España, Ministerio de Medio Ambiente y Medio Rural y Marino & Instituto de Meteorologia de Portugal, Madrid/Lisboa.

Aronson, J., Pereira, J.S., Pausas, J.G., 2009. Introduction, in: Aronson, J., Pereira, J.S., Pausas, J.G. (Eds.), Cork Oak Woodlands on the Edge: Ecology, Adaptive Management and Restoration. Island Press, Washington, D.C., USA, pp. 1-6.

Auld, G., Gulbrandsen, L.H., McDermott, C.L., 2008. Certification schemes and the impacts on forests and forestry. Ann. Rev. Environ. Resour. 33, 187–211.

Autoridade Florestal Nacional, 2010. Final Report of the 5th National Forest Inventory. Autoridade Florestal Nacional, Lisboa.

Belsky, A.J., Matzke, A., Uselman, S., 1999. Survey of livestock influences on stream and riparian ecosystems in the western United States. J. Soil Water Conserv. 54, 419–431.

Bennett, E.L., 2001. The joint effort of timber certification. Conserv. Biol. 15, 318–319.

Berrahmouni, N., Regato, P., Ellatifi, M., Daly-Hassen, H., Bugalho, M.N., Bensaid, S., Diaz, M., Aronson, J., 2009. Ecoregional Planning for Biodiversity Conservation, in: Aronson, J., Pereira, J.S., Pausas, J.G. (Eds.), Cork Oak Woodlands on the Edge: Ecology, Management, and Restoration. Island Press, Washington, D.C., USA, pp. 203–217.

Borchers, D.L., Buckland, S.T., Goedhart, P.W., Clarke, E.D., Hedley, S.L., 1998. Horvitz-Thompson estimators for double-platform line transect surveys. Biometrics 54, 1221-1237.

Buckland, S.T., Anderson, D.R., Burnham, K.P., Laake, J.L., Borchers, D.L., Thomas, L., 2001. Introduction to Distance Sampling: Estimating Abundance of Biological Populations. Oxford University Press, USA.

Bugalho, M.N., Caldeira, M.C., Pereira, J.S., Aronson, J., Pausas, J.G., 2011. Mediterranean cork oak savannas require human use to sustain biodiversity and ecosystem services. Front. Ecol. Environ 9, 278–286.

Bugalho, M.N., Lecomte, X., Gonçalves, M., Caldeira, M.C., Branco, M., 2011. Establishing grazing and grazing-excluded patches increases plant and invertebrate diversity in a Mediterranean oak woodland. For. Ecol. Manage. 261, 2133–2139.

Bugalho, M.N., Plieninger, T., Aronson, J., Ellatifi, M., Crespo, D.G., 2009. Open Woodlands: A Diversity of Uses (and Overuses), in: Aronson, J., Pereira, J.S., Pausas, J.G. (Eds.), Cork Oak Woodlands on the Edge: Ecology, Adaptive Management and Restoration. Island Press, Washington, D.C., USA, pp. 33–47.

Bugalho, M.N., Dias, F.S., Briñas, B., Cerdeira, J.O., 2015. Using the high conservation value forest concept and Pareto optimization to identify areas maximizing biodiversity and ecosystem services in cork oak landscapes. Agrofor. Syst. 1–10

Caldeira, M.C., Ibáñez, I., Nogueira, C., Bugalho, M.N., Lecomte, X., Moreira, A., Pereira, J.S., 2014. Direct and indirect effects of tree canopy facilitation in the recruitment of Mediterranean oaks. J. Appl. Ecol. 51, 349–358.

Campos, P., Daly-Hassen, H., Ovando, P., 2007. Cork oak forest management in Spain and Tunisia: two case studies of conflicts between sustainability and private income. Int. For. Rev. 9, 610–626.

Canfield, R.H., 1941. Application of the Line Interception Method in Sampling Range Vegetation. J. For. 39, 388–394.

Caparrós, A., Huntsinger, L., Oviedo, J.L., Plieninger, T., Campos, P., 2013. Economics of Ecosystem Services, in: Campos, P., Huntsinger, L., Pro, J.L.O., Starrs, P.F., Diaz, M., Standiford, R.B., Montero, G. (Eds.), Mediterranean Oak Woodland Working Landscapes. Springer Netherlands, pp. 353–388.

Cardoso, J.C., 1974. A classificação dos solos de Portugal – Nova versão. Bol. Solos S.R.O.A. 17, 14–46.

Chaideftou, E., Thanos, C.A., Bergmeier, E., Kallimanis, A., Dimopoulos, P., 2009. Seed bank composition and above-ground vegetation in response to grazing in sub-Mediterranean oak forests (NW Greece). Plant Ecol. 201, 255–265.

Dias, F.S., Bugalho, M.N., Rodríguez-González, P.M., Albuquerque, A., Cerdeira, J.O., 2015. Effects of forest certification on the ecological condition of Mediterranean streams. J. Appl. Ecol. 52, 190–198.

Díaz, M., Campos, P., Pulido, F.G., 1997. The Spanish dehesas: a diversity of land uses and wildlife, in: Pain, D., Penkowski, M. (Eds.), Farming and Birds in Europe:

The Common Agricultural Policy and Its Implications for Bird Conservation. Academic Press, London, pp. 178–209.

Díaz, M., Tietje, W.D., Barrett, R.H., 2013. Effects of Management on Biological Diversity and Endangered Species, in: Campos, P., Huntsinger, L., Pro, J.L.O., Starrs, P.F., Diaz, M., Standiford, R.B., Montero, G. (Eds.), Mediterranean Oak Woodland Working Landscapes. Springer Netherlands, pp. 213–243.
Fox, J., Weisberg, H.S., 2010. An R Companion to Applied Regression, 2nd ed. SAGE Publications, Thousand Oaks, California.

FSC, 2014. Global FSC certificates: type and distribution. Forest Stewardship Council, Bonn, Germany.

Gómez-Aparicio, L., Gómez, J.M., Zamora, R., Boettinger, J.L., 2005. Canopy vs. soil effects of shrubs facilitating tree seedlings in Mediterranean montane ecosystems. J. Veg. Sci. 16, 191–198.

Gulbrandsen, L.H., 2010. Transnational Environmental Governance: The Emergence and Effects of the Certification of Forest and Fisheries. Edward Elgar Publishing, UK

Gullison, R.E., 2003. Does forest certification conserve biodiversity? Oryx 37, 153–165.

Hedley, S.L., Buckland, S.T., 2004. Spatial models for line transect sampling. J. Agric. Biol. Environ. Stat. 9, 181–199.

Hernández, E.I., Vilagrosa, A., Pausas, J.G., Bellot, J., 2010. Morphological traits and water use strategies in seedlings of Mediterranean coexisting species. Plant Ecol. 207, 233–244.

Huntsinger, L., Campos, P., Starrs, P.F., Oviedo, J.L., Díaz, M., Standiford, R.B., Montero, G., 2013. Working Landscapes of the Spanish Dehesa and the California Oak Woodlands: An Introduction, in: Campos, P., Huntsinger, L., Pro, J.L.O., Starrs, P.F., Diaz, M., Standiford, R.B., Montero, G. (Eds.), Mediterranean Oak Woodland Working Landscapes. Springer Netherlands, pp. 3–23.

Krebs, C.J., 1998. Ecological Methodology. Benjamin Cummings, Menlo Park, California

Marques, T.A., Thomas, L., Fancy, S.G., Buckland, S.T., 2007. Improving estimates of bird density using multiple covariate distance sampling. The Auk 127.

Marra, G., Wood, S.N., 2011. Practical variable selection for generalized additive models. Comput. Stat & Data Anal. 55, 2372–2387.

METI, NASA, 2011. ASTER Global Digital Elevation Model (ASTER GDEM). Ministry of Economy, Trade and Industry of Japan (METI) and National Aeronautics and Space Administration (NASA), Japan/USA.

Miller, D.L., Rexstad, E., Burt, L., Bravington, M.V., Hedley, S., 2015. dsm: density surface modelling of distance sampling data, R package version 2.2.9, Available at: http://CRAN.R-project.org/package=dsm

Miller, D.L., Burt, M.L., Rexstad, E.A., Thomas, L., 2013. Spatial models for distance sampling data: recent developments and future directions. Methods Ecol. Evol. 4, 1001–1010

Myers, N., Mittermeier, R.A., Mittermeier, C.G., Fonseca, G.A.B., Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature 403, 853–858.

Navarro-González, I., Pérez-Luque, A.J., Bonet, F.J., Zamora, R., 2013. The weight of the past: land-use legacies and recolonization of pine plantations by oak trees. Ecol. Appl. 23, 1267–1276.

Pausas, J.G., Marañon, T., Caldeira, M., Pons, J., 2009. Natural Regeneration, in: Aronson, J., Pereira, J.S., Pausas, J.G. (Eds.), Cork Oak Woodlands on the Edge: Ecology, Adaptive Management and Restoration. Island Press, Washington, D.C., USA.

Pausas, J., Pereira, J.S., Aronson, J., 2009. The Tree, in: Aronson, J., Pereira, J.S., Pausas, J.G. (Eds.), Cork Oak Woodlands on the Edge: Ecology, Management, and Restoration. Island Press, Washington, D.C., USA, pp. 7–21.

PEFC, 2014. PEFC Global Statistics: SFM & CoC Certification. Program for the Endorsement of Forest Certification, Geneva, Switzerland.

Pinheiro, J., Bates, D., 2009. Mixed-Effects Models in S and S-PLUS. Springer, New York, USA.

Plieninger, T., 2007. Compatibility of livestock grazing with stand regeneration in Mediterranean holm oak parklands. J. Nat. Conserv. 15, 1–9.

Plieninger, T., Rolo, V., Moreno, G., 2010. Large-Scale Patterns of *Quercus ilex*, *Quercus suber*, and *Quercus pyrenaica* Regeneration in Central-Western Spain. Ecosystems 13, 644–660.

Plieninger, T., Schaich, H., Kizos, T., 2011. Land-use legacies in the forest structure of silvopastoral oak woodlands in the Eastern Mediterranean. Reg. Environ. Change 11, 603–615

Pollard, J.H., 1971. On distance estimators of density in randomly distributed forests. Biometrics 27, 991.

Pons, J., Pausas, J.G., 2006. Oak regeneration in heterogeneous landscapes: The case of fragmented Quercus suber forests in the eastern Iberian Peninsula. For. Ecol. Manage. 231, 196–204.

Puerta-Piñero, C., Gómez, J.M., Valladares, F., 2007. Irradiance and oak seedling survival and growth in a heterogeneous environment. For. Ecol. Manage. 242, 462–469.

Pulido, F., García, E., Obrador, J.J., Moreno, G., 2010. Multiple pathways for tree regeneration in anthropogenic savannas: incorporating biotic and abiotic drivers into management schemes: Pathways for oak recruitment in savannas. J. Appl. Ecol. 47, 1272–1281.

Putz, F.E., Zuidema, P.A., Synnott, T., Peña-Claros, M., Pinard, M.A., Sheil, D., Vanclay, J.K., Sist, P., Gourlet-Fleury, S., Griscom, B., Palmer, J., Zagt, R., 2012. Sustaining conservation values in selectively logged tropical forests: the attained and the attainable: Sustaining tropical forests with forestry. Conserv. Lett. 5, 296–303.

QGIS Development Team, 2014. QGIS Geographic Information System. Open Source Geospatial Foundation Project. http://qgis.osgeo.org.

Ramírez, J.A., Díaz, M., 2008. The role of temporal shrub encroachment for the maintenance of Spanish holm oak *Quercus ilex* dehesas. For. Ecol. Manage. 255, 1976–1983.

R Core Team, 2014. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

Rey Benayas, J.M., Bullock, J.M., Newton, A.C., 2008. Creating woodland islets to reconcile ecological restoration, conservation, and agricultural land use. Front. Ecol. Environ 6, 329–336.

Rodwell, J.S., 1998. British Plant Communities. Cambridge University Press.

Serrasolses, I., Caldeira, M., Vilagrosa, A., Pausas, J.G., Sauras, T., Cortina, J., Vallejo, V.R., 2009. Soil Properties Constraining Cork Oak Distribution, in: Aronson, J., Pereira, J.S., Pausas, J.G. (Eds.), Cork Oak Woodlands on the Edge: Ecology, Adaptive Management and Restoration. Island Press, Washington, DC, USA.

Thomas, L., Buckland, S.T., Rexstad, E. a, Laake, J.L., Strindberg, S., Hedley, S.L., Bishop, J.R., Marques, T. a, Burnham, K.P., 2010. Distance software: design and analysis of distance sampling surveys for estimating population size. J. Appl. Ecol. 47, 5–14.

Tollefson, C., Gale, F., Haley, D., 2009. Setting the Standard: Certification, Governance, and the Forest Stewardship Council. Univ of British Columbia Press, Vancouver B.C.

Tyler, C.M., Kuhn, B., Davis, F.W., 2006. Demography and recruitment limitations of three oak species in California. Q Rev Biol 81, 127–152.

Veldman, J.W., Buisson, E., Durigan, G., Fernandes, G.W., Le Stradic, S., Mahy, G., Negreiros, D., Overbeck, G.E., Veldman, R.G., Zaloumis, N.P., Putz, F.E., Bond, W.J., 2015. Toward an old-growth concept for grasslands, savannas, and woodlands. Front. Ecol. Environ 13, Venables, W.N., 2003. Modern Applied Statistics with S, 4th edition. ed. Springer, New York.

Weltzin, J.F., McPherson, G.R., 1999. Facilitation of conspecific seedling recruitment and shifts in temperate savanna ecotones. Ecol. Monogr. 69, 513–534.

Wendland, K.J., Honzák, M., Portela, R., Vitale, B., Rubinoff, S., Randrianarisoa, J., 2010. Targeting and implementing payments for ecosystem services: Opportunities for bundling biodiversity conservation with carbon and water services in Madagascar. Ecol. Econ. 69, 2093–2107.

Wood, S., 2006. Generalized Additive Models: An Introduction with R, 1st ed. Chapman and Hall/CRC, Boca Raton, FL.

Wood, S.N., 2013. A simple test for random effects in regression models. Biometrika 100, 1005–1010.

Wood, S.N., 2011. Fast stable restricted maximum likelihood and marginal likelihood estimation of semiparametric generalized linear models. J. R. Stat. Soc.: Series B (Statistical Methodology) 73, 3–36.

Zuur, A., Ieno, E.N., Walker, N., Saveliev, A.A., Smith, G.M., 2009. Mixed Effects Models and Extensions in Ecology with R. Springer, New York, USA.

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

Estate	Years of certification	# CZ	Mean area of CZ (ha)	# Transects on CZ	# NCZ	Mean area of NCZ (ha)	# Transects on NCZ
Arrão de Baixo	6	1	44.78	3	1	44.78	3
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 do Grou	6	2	100.54	9	2	100.54	8
Onzenas de cima	6	1	5.60	3	1	5.60	3
Pereira	6	1	8.49	3	1	8.49	3
TOTAL		14		45	13		41

Table 2 – List of the variables included in the local and topographic density surface 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".
Local variables	Absolute cover of adult cork oaks	m^2/ha
	Shrub richness	Number of species

	Shrub diversity	Shannon-Weaver diversity index
	Lavandula stoechas cover	% of cover
	Rosmarinus officinalis cover	% of cover
	Cistus salvifolius cover	% of cover
	Ulex spp. cover	% of cover by *Ulex minor* and *Ulex australis*
	Cistus crispus cover	% of cover
	Cistus monspeliensis cover	% of cover
	Altitude	Metres
	Slope	Percentage
	Aspect	Azimuth
	Topographic ruggedness index	
	Topographic position index	
Topographic variables	Latitude	
	Longitude	
	Litholic soils	Percentage of cover
	Clayey soils with low saturation	Percentage of cover
	Podzol soils	Percentage of cover

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

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 (+), *Ulex sp* cover, *R. officinalis* cover,	59.4%

			s(Estate)	
	Young trees	Tweedie	s(Shrub diversity), s(Estate)	48.9%
Topographic	Seedlings	Tweedie	Zone type:conservation zone (+), s(Estate)	24.4%
	Saplings	Tweedie	Zone type:conservation zone (+), s(x and y coordinates)	44.7%
	Young trees	Poisson	s(Aspect), s(Area of litholitic non humic soil), s(Area of podzol soils), s(Estate)	63.1%

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

Variable	p-value	Spatial correlation structure
Species richness	**0.004 (+)**	Gaussian
Shannon- -Weaver diversity index	**0.008 (+)**	Gaussian

	Overall	0.448	
	Rosmarinus officinalis	0.790	
	Cistus salvifolius	0.318	
	Ulex sp	**0.010 (+)**	Gaussian
	Cistus crispus	0.919	
	Lavandula stoechas	0.645	
Shrub cover	*Cistuas monspelliensis*	0.161	
	Hallimium lasianthum	0.299	
	Cistus ladanifer	0.101	
	Daphne gnidium	0.628	
	Quercus lusitanica	0.455	
	Pterospartum tridentatum	0.180	
	Rubia peregrina	0.125	

	Hallimium ocymoides	0.317	
	Erica australis	0.363	
	Pistacia lentiscus	0.298	
	Myrtus communis	0.312	

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”.

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

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

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

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).

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

Figure 1

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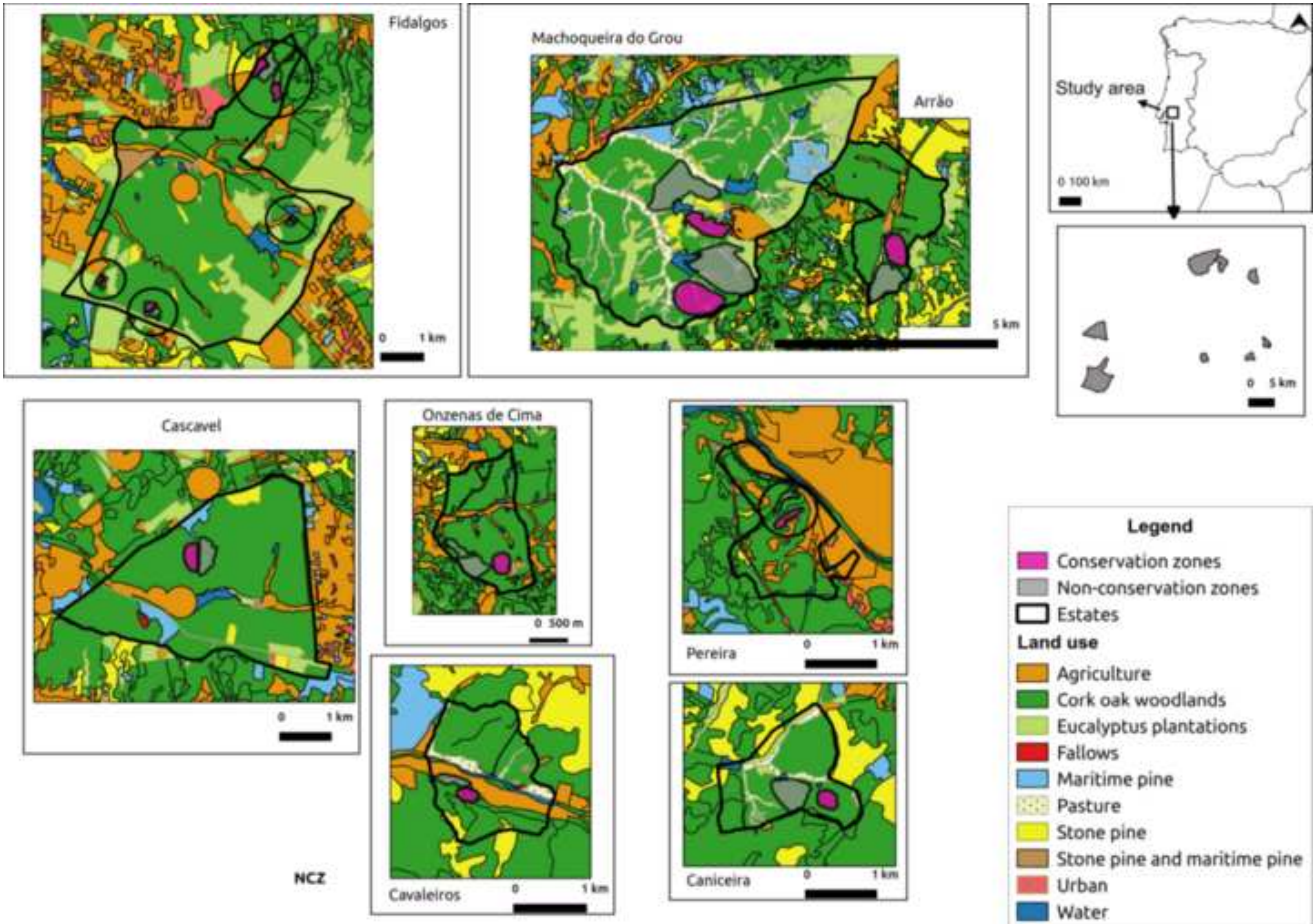

Figure 2
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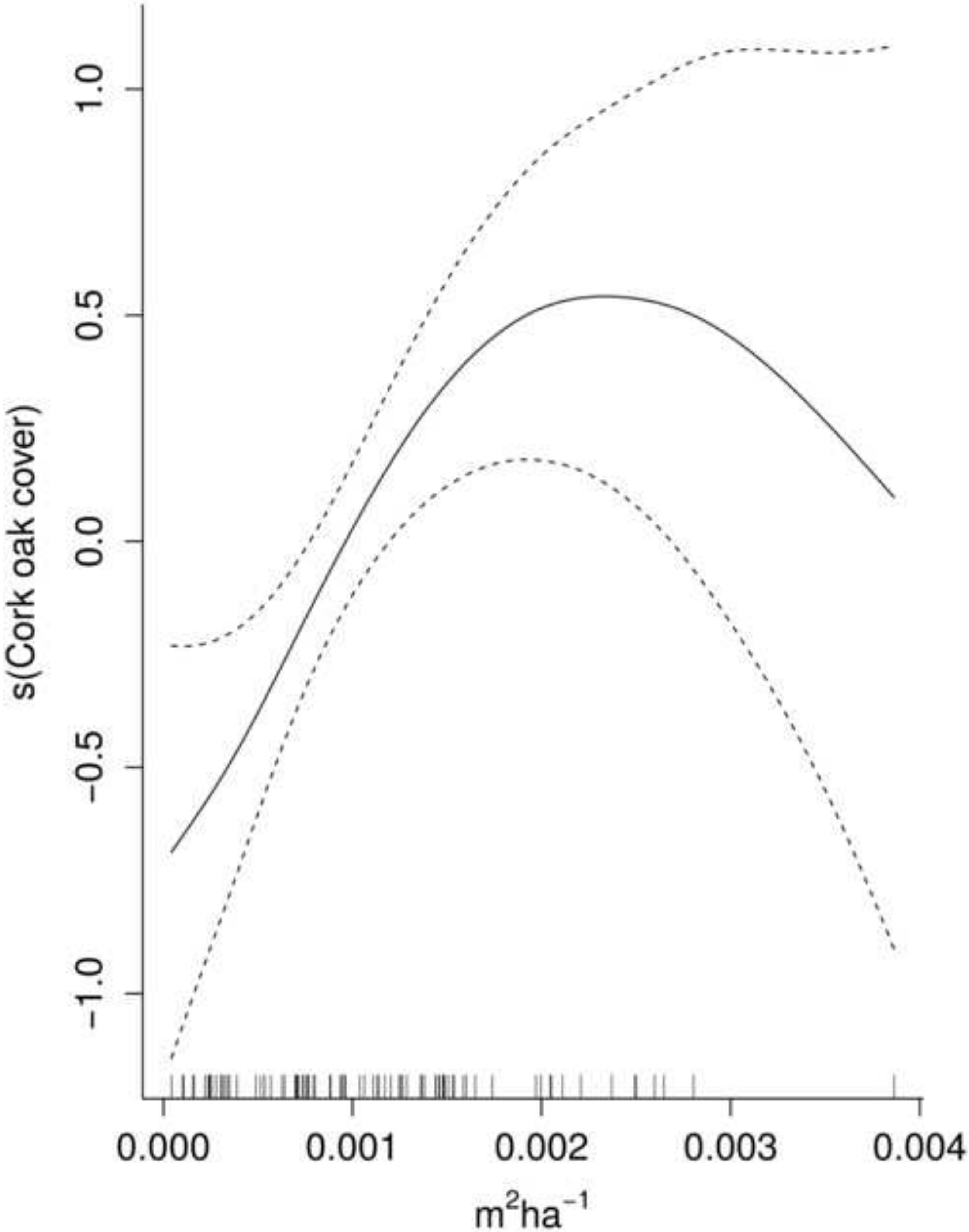

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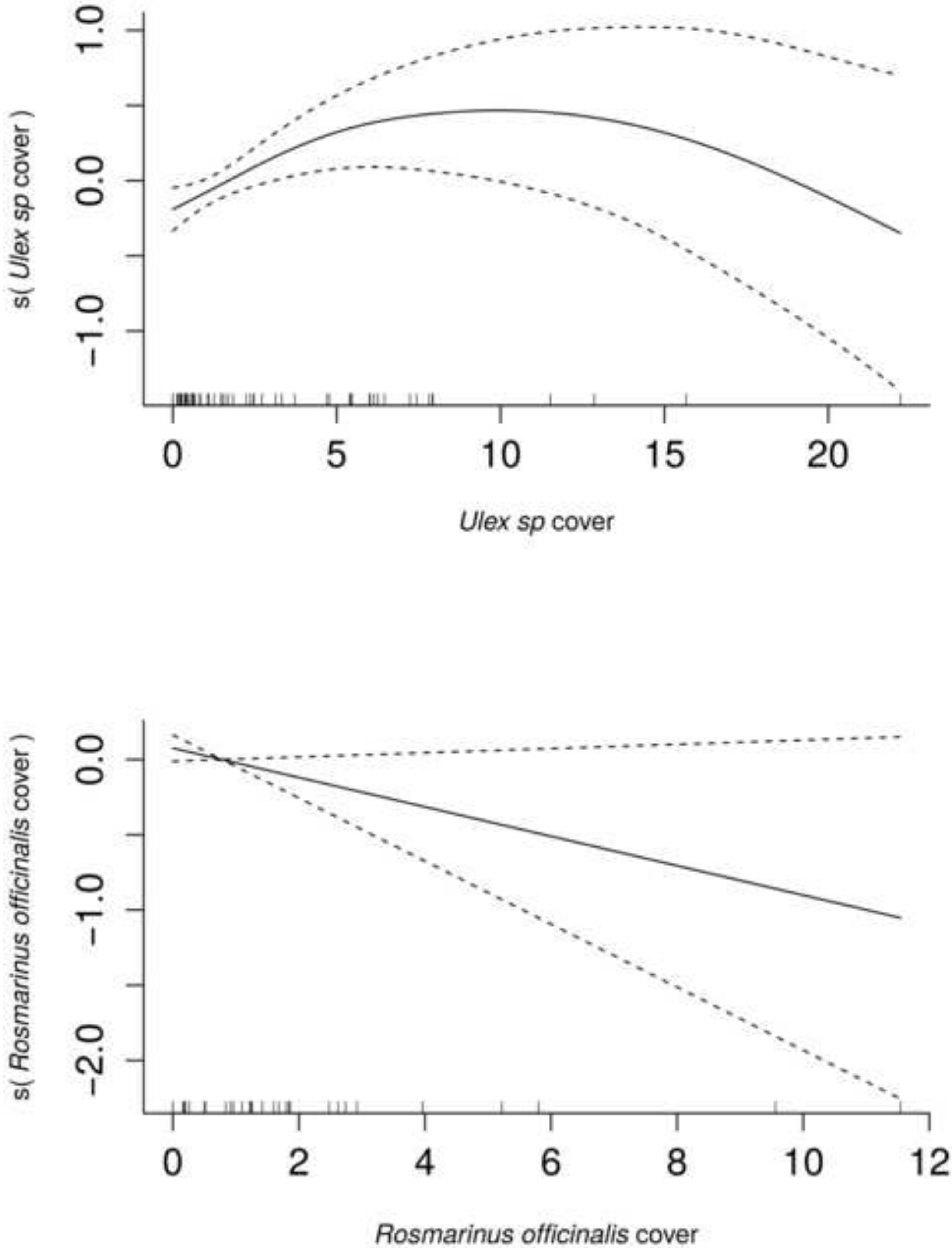

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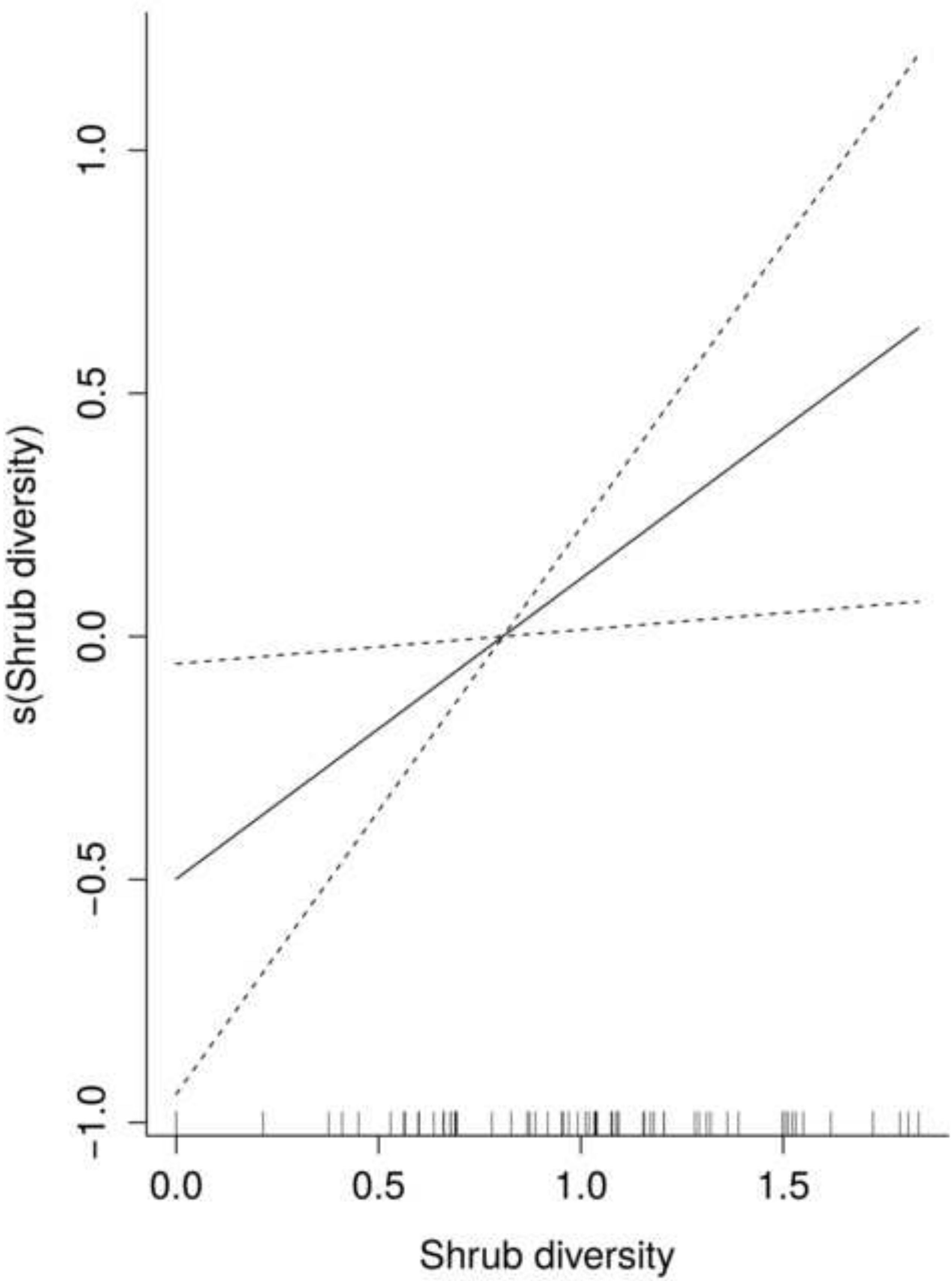

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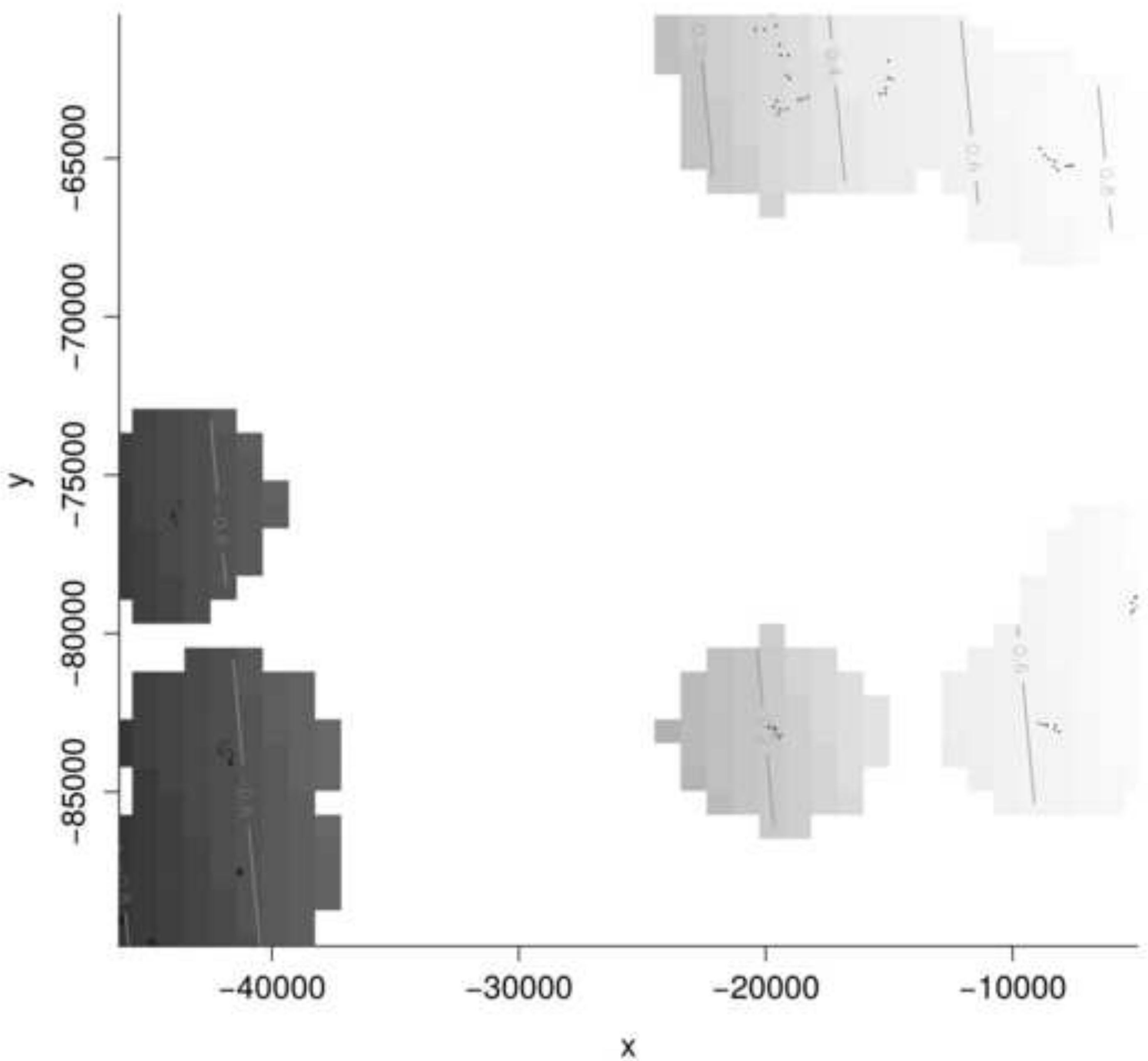

Figure 6
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