Ecological Modelling

Long-term sustainability of cork oak agro-forests in the Iberian Peninsula: A model-based approach aimed at supporting the best management options for the *montado* conservation

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ABSTRACT

The future of the *montado*, a human shaped agro-forestry ecosystem of South Western Europe, is questioned due to the observed lack of cork oak health and low natural regeneration. We developed a System Dynamics Modelling approach to predict the long-term sustainability of this agro-forest, by recreating cork-oak population dynamics, management practices and the main environmental and biological constrains associated with this ecosystem. Our results indicate that the leading limitations to cork oak regeneration in *montado* ecosystems result from the intensity and interaction of land management practices, namely livestock and the use of heavy machinery. The main conclusions indicate that limiting the quantity of livestock up to 0.40 LU.ha^{-1}, and considering soil ploughing with a minimum periodicity of 5 years, are crucial to maintaining sustainable cork oak populations. This study represents a first step to support strategic options for cork oak *montado* management by providing projections of long-term population trends under realistic social-ecological change scenarios.

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1. Introduction

The largest cork oak woodlands are found in the Iberian Peninsula, where cork harvesting, agricultural, pastoral and other traditional uses have been practiced at least since the Middle Ages (Joffre et al., 1999; Olea et al., 2005; Bugalho et al., 2011). This agro-silvo-pastoral ecosystem, known as *montado* or *dehesa* (the Portuguese and Spanish names, respectively), is composed typically of open woodlands (20–80 trees ha^{-1}) with only one or a few tree species such as cork oak (*Quercus suber*), holm oak (*Quercus rotundifolia*) and pines (*Pinus* spp.) (Joffre et al., 1999; Pinto-Correia and Mascarenhas, 1999; Pinto-Correia and Fonseca, 2009). Moreover, the *montado* ecosystem is considered both a protected habitat within the EU habitats directive (92/43/EEC) and a high nature value farming system, according to the classification proposed by the European Environmental Agency (Paracchini et al., 2008).

The dominant tree of the *montado*, the cork oak (*Quercus suber*), is a sclerophyllous evergreen oak that occurs in non-carbonated soils (Pausas et al., 2009) restricted to the western part of the Mediterranean Basin (Tutin et al., 1964). This is a slow-growing tree with a high longevity like other oaks, with a lifespan of 250–300 years. Earliest flowering and fructification occur at around 15–20 years of age, producing both annual and biennial acorns (Natividade, 1950; Elena-Rosselló et al., 1993; Díaz-Fernández et al., 2004; Pereira and Tomé, 2004). Acorns can be dispersed by animals or predated. The main seed dispersers are wood mouse (*Apodemus sylvaticus*) and European jay (*Garrulus glandarius*), which are involved in short- and long-distance dispersal, respectively (Herrera, 1995; Pausas et al., 2009). Acorn predators are mainly seed-boring insects, such as the acorn weevil (*Curculio* spp., Coleoptera) and the acorn moth (*Cydia* spp., Lepidoptera) (Ceia and Ramos, 2014). In open areas, predators can also be livestock, deer, wild boar, birds and rabbits whereas under a shrub canopy acorn predation is mostly carried out by small rodents, since neither large mammals nor birds will penetrate the dense vegetation (Herrera, 1995). Cork oak trees grow in areas with strong seasonal water deficits by surviving to droughts in part due to their
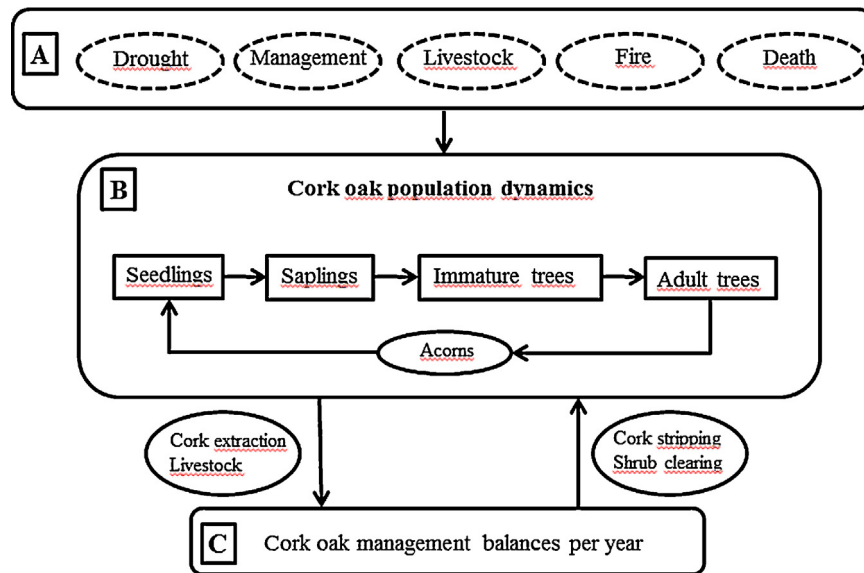
Fig. 1. Conceptual diagram of the dynamic model used to: B) simulate the cork oak population dynamics, assuming cork oak age stratification (seedlings, saplings, immature trees and adult trees), A) simulate the associated environmental constrains and C) the respective economic balance sub-model.

extensive and deep root system and in some cases seedlings shed leaves and resprout when the drought is over (Acácio et al., 2007; Gómez-Aparicio et al., 2008; Arosa et al., 2015). The cork oak has a thick insulating bark which evolved as a fire protection mechanism (Pausas, 1997; Catry et al., 2012) and which regrows after extraction for cork production (Pausas et al., 2009). Cork extraction represents an important resource of high economic value in the Iberian Peninsula as cork thickness and properties make it valuable raw material for industry. After the removal of the outer bark, a traditional practice that does not harm the tree and occurs every 9 years, the tree has the capacity to produce a new cork bark by adding new layers of cork every year and this may be repeated throughout the tree lifetime (Pausas et al., 2009). The annual production of cork is about 370 000 t, mostly from the cork oaks of Portugal and Spain that produce respectively 51% and 23% of the world total (Pereira and Tomé, 2004).

Our main goal was the integration of disperse information from specific studies concerning the different life stages and stressors in a single user-friendly platform for understanding and predicting the future sustainability of the *montado* ecosystem. We considered as stressors the lack of cork oak health and the low natural regeneration rates. Tree health has been affected by intensive pruning, exaggerated cork harvesting and the influence of pests and diseases (Camilo-Alves et al., 2013; Acácio and Holmgren, 2014). The limitations to natural regeneration have been attributed to various causes, including: a) poor dispersal and shortage of viable acorns (Branco et al., 2002; Pulido and Díaz, 2005; Acácio et al., 2007); b) high post-dispersive acorn losses associated with seedling mortality due to over-grazing by livestock and wild animals (Herrera, 1995; Plieninger et al., 2004; Acácio et al., 2007; Pulido et al., 2013); c) low seedling survival to summer drought (Gimeno et al., 2009; Smit et al., 2009). Besides, the development of farm mechanization, including the generalised use of wide plows, disc harrows, and scarifiers destroys young trees and may damage roots and weaken established trees, creating more susceptibility to the attack of pests and diseases (Branco and Ramos, 2009; Arosa et al., 2015). The fire frequency detected in the last few decades in this ecosystem (Pausas and Vallejo, 1999) has also become an additional obstacle to cork oak survival and recruitment (Acácio et al., 2009, 2010). Although cork oak is able to exhibit fire-resistance characteristics due to the bark insulation and to the mechanism of resprout-

ing afterwards, frequent or intense wildfires may kill adult trees, especially if these events occur immediately after cork extraction (Moreira et al., 2007). Additionally, as consequence of concomitant change of mean annual temperature and rainfall extremes during the last decades, the recurrent droughts will reduce the canopy cover and limit cork oak regeneration (Vivas and Maia, 2008; Acácio et al., 2009).

Considering its ecological and economic sustainability, the conservation of these ecosystems requires the comprehension of its capacity to withstand and recover from disturbances imposed by natural and anthropogenic factors (Müller, 2005; Kandziora et al., 2013; Stoll et al., 2015). Ecological modelling provides useful tools to study complex systems by predicting the outcome of alternative scenarios, and might help guiding the most correct management options from projected future outcomes (e.g. Bastos et al., 2012, 2015; Fernandes et al., 2013; Santos et al., 2013). Actually dynamic models can be used to support the mechanistic understanding of complex multifactorial ecological processes as they simultaneously integrate the structure and the composition of systems for a specific period (Jørgensen, 1994, 2001). When properly developed, tested and applied with insight and with respect for their underlying assumptions, dynamic models are capable of simulating conditions that are difficult or impossible to produce otherwise (Jørgensen, 2001).

We have developed a System Dynamics Modelling approach to recreate the management practices associated with cork oak *montado* and the main biological and environmental drivers influencing this semi-natural ecosystem, in order to assess its long-term sustainability (Sterman, 2001). This study is the first to integrate in a single approach the multifactorial factors associated with cork oak population dynamics. Specifically, our objectives were: (1) integrating in a user-friendly platform the biotic and abiotic factors, acting as a whole, with holistic and interrelated influences on the cork oak population dynamics (2) simulate realistic scenarios in order to understand the most determinant factors implicated in the cork oak decline across the Iberian Peninsula and (3) highlight the most effective management practices through quantitative metrics with implications for the long-term sustainability and conservation of the *montado* ecosystem.
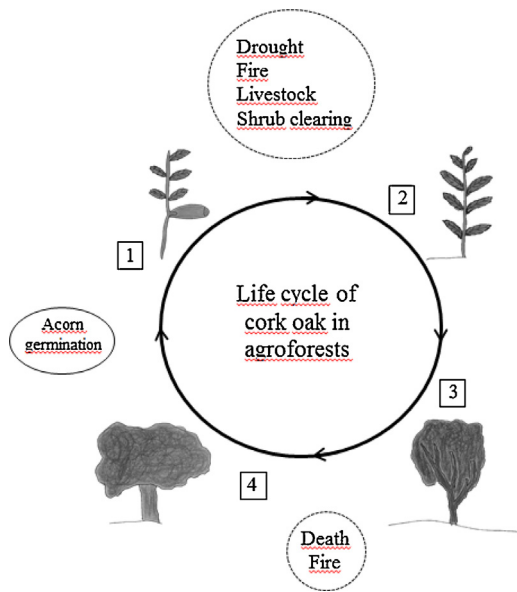
Fig. 2. Classes in the life cycle of cork oak in agroforests considered for the design of the conceptual diagram of the four sub-models: *1* seedlings, *2* saplings, *3* immature trees, *4* adult trees and the processes influencing each cohort of individuals. Drought, fire, livestock and shrub clearing are processes that most affect *1* and *2* classes. Death and fire have an important effect over *3* and *4* classes. Individuals of a specific class transit to the next when completing their development.

2. Methods

2.1. Dynamic model conceptualization

Cork oak population dynamics in agro-forests were modelled to anticipate the potential effects of current management practices under different environmental scenarios expected for the 21st century. Natural regeneration is a dynamic process where new individuals are recruited into the mature population, compensating the losses due to mortality, induced by several biotic and abiotic factors (Harper, 1977; Holmgren et al., 1997; Pulido and Díaz, 2005; Fei and Steiner, 2008). Therefore, we developed a System Dynamics Modelling approach composed by four sub-models concerning the cork oak population dynamics based on cork oak age stratification (classes) (González et al., 2015), represented by the sum of different life stage cohorts (seedlings, saplings, immature trees and adult trees), and one additional sub-model to estimate the economic balances associated (Fig. 1).

The year was chosen as time unit and the simulation period was established for 100 years, considered suitable to capture the influence of main factors on the long term cork oak population dynamics, namely those induced by management options. All modelling was performed with the software STELLA, version 10.0.5 (Isee Systems, Inc.). The original conceptual diagram of the overall sub-models (Appendix I in Supplementary material), variables and equations included in the model construction (Table 1 and Appendix II in Supplementary material) and full explanation of processes (Appendix III in Supplementary material), are available as supplementary electronic material.

3. Natural regeneration

In order to recreate cork oak natural regeneration, four sub-models (seedlings, saplings, immature trees, adult trees) were designed to reproduce the most relevant classes, dynamics and environmental constrains associated to the cork oak populations (Fig. 1, Table 1, Appendix I, Appendix II in Supplementary material, Fig. 3). For each sub-model, cork oak trees were grouped into classes, considering their stage, size and diameter at breast height (DBH): seedlings (acorn or cotyledon scatters still attached, height <50 cm), saplings (height > 50 cm, DBH < 10 cm), immature tree (height >50 cm, DBH > 10 cm) and adult tree (DBH > 25 cm) (Muick and Bartolome, 1987; Montero and Cañellas, 2003). Transitions between classes and within each class were affected by biotic and abiotic factors that implicate cause-effect relationships. Data used in the model was compiled from publications regarding the cork oak' life cycle and the cork oak *montado* characterization in the Iberian Peninsula, ranging from inland regions, with lower annual precipitation (550 mm) and higher temperature (17 °C), to coastal regions, with higher rainfall (1 800 mm) and milder temperatures (14 °C) (Table 1).

3.1. Biotic and abiotic processes influencing cork oak natural regeneration

Several biotic and abiotic processes were assumed as influent drivers on the cork oak population dynamics. The seedlings and saplings were mainly affected by droughts, fire effects, livestock and grazing pressure, and shrub clearing intensity (Fig. 2). On the other hand, immature and adult trees were affected by fire and natural death by aging. Besides the effect of external factors, the transition of individuals throughout classes considered into the model is also limited by intraspecific competition and by the success of established cork oak plants.

3.1.1. Acorn germination

To model the germination of acorns, each adult tree was assumed to produce between 1.32 and 33.6 kg of acorns per year (Herrera, 1995; Cañellas et al., 2007; Pérez-Ramos et al., 2008). Since the germination of acorns is limited by biotic factors related with predation and insect damages, the effective number of acorns supplying the system was dependent on the predation/survival rate and of the proportion of viable acorns (Branco et al., 2002; Acacio et al., 2007). Furthermore, acorns that do not germinate and remain in the understory dry out and become inviable before the beginning of a new year.

3.1.2. Competition

Cork oak competition in *montado* involves an intraspecific struggle for basic resources, such as light, water and/or nutrients (Flores-Martinez et al., 1994; Holmgren et al., 1997; Callaway and Walker, 1997). Our dynamic model was designed to recreate intraspecific competition within and between the different classes, from the sapling to the adult trees. During the tree development, competition varies from soil resources to light, this last affecting the shape and size of crowns (Kenkel, 1988; Deleuze et al., 1996). With increasing tree cover and light limitation, the older individuals begin to disproportionately obstruct the incoming solar radiation because of its unidirectional nature, thereby suppressing the growth of younger trees (Dolezal et al., 2004). In this context, Assmann and Gardiner (1970) defined that the area available for a tree is represented by its crown projection versus the corresponding fraction not occupied. Serrada (2002) estimated that this method suffers the disadvantage of supposing a full canopy cover (100%) but in cork oak *montados* the normal canopy cover is incomplete, thus we used the Di Bérengerís method for calculating average crown projection using the diameter at breast height (DBH). This method enables the calculation of tree distribution within an area, assuming a mixture of trees belonging to all classes. Through the function $y = 0.0431 \bullet x^{1.6025}$ (x is DBH in cm) the maximum cork oak density (trees ha^{-1}) belonging to each size class was limited in the model (Natividade, 1950; Marín-Pageo and Camacho, 2011).

Table 1
Specification of the main variables included in the model, respective description, measure units and references.

Cork oak population dynamics			
Variable	Description	Unit	Source
Seedlings			
Random acorns per tree	Average kg of acorns per tree Minimum acorns per tree = 1.32 Maximum acorns per tree = 33.6	kg of acorns	Herrera (1995), Cañellas et al. (2007), Pérez-Ramos et al. (2008)
Random acorns Predation rate	Proportion of predated acorns before germination Minimum acorns predation rate = 0.36 Maximum acorns predation rate = 0.83	rate	Acácio et al. (2007)
Random acorns insect damage rate	Proportion of acorns damaged by insects before germination Minimum acorns insect damage rate = 0.09 Maximum acorns insect damage rate = 0.68	rate	Branco et al. (2002), Acácio et al. (2007)
Random acorns Acorn germination rate	Proportion of acorns successfully germinated Minimum acorns germination rate = 0.38 Maximum acorns germination rate = 0.84	rate	Herrera (1995), Acácio et al. (2007), Arosa et al. (2015)
Average seedlings fire mortality rate	Proportion of seedling mortality by fire Fire seedlings mortality = 0.0008	rate	Catry et al. (2012)
Random seedling drought mortality rate	Proportion of seedling mortality by drought Minimum seedlings drought mortality rate = 0.68 Maximum seedlings drought mortality rate = 0.70	rate	Arosa et al. (2015)
Random seedlings livestock mortality rate	Proportion of seedling mortality by livestock Minimum seedlings livestock mortality rate = 0.28 Maximum seedlings livestock mortality rate = 0.65	rate	Plieninger et al. (2004)
Random seedlings shrub clearing mortality rate	Proportion of seedling mortality by shrub clearing Minimum seedlings shrub clearing mortality rate = 0 Maximum seedlings shrub clearing mortality rate = 1	rate	Arosa et al. (2015)
Average seedlings resprouting rate	Proportion of seedling successfully resprouted = 0.239	rate	Arosa et al. (2015)
Saplings			
Average saplings fire mortality rate	Proportion of saplings mortality by fire Fire saplings mortality = 0.0008	rate	Catry et al. (2012)
Random saplings cohort × drought mortality rate	Proportion of saplings mortality by drought Minimum saplings drought mortality rate = 0.68 Maximum saplings drought mortality rate = 0.70	rate	Arosa et al. (2015)
Random saplings cohort × livestock mortality rate	Proportion of saplings mortality by livestock Minimum saplings livestock mortality rate = 0.57 Maximum saplings livestock mortality rate = 0.87	rate	Plieninger et al. (2004)
Random saplings cohort × shrub clearing mortality rate	Proportion of saplings mortality by shrub clearing Minimum saplings shrub clearing mortality rate = 0 Maximum saplings shrub clearing mortality rate = 1	rate	Arosa et al. (2015)
DBH saplings	Saplings average diameter at breast height = 6	cm	Pulido et al. (2013)
Immature trees			
Random immature trees cohort × fire mortality rate	Proportion of immature trees mortality by fire Minimum immature trees cohort × fire mortality rate = 0.06 Maximum immature trees cohort × fire mortality rate = 0.08	rate	Catry et al. (2012)
Random immature trees cohort × death rate	Proportion of immature trees death Minimum immature trees cohort × death rate = 0 Maximun immature trees cohort × death rate = 0.000092	rate	Ribeiro and Surový (2008)
DBH immature trees	Immature trees average diameter at breast height = 10	cm	Pulido et al. (2013)
Adult trees			
Adult trees per ha	Initial density of adult trees = 50	trees ha^{-1}	Joffre et al. (1999)
Random adult trees fire mortality rate	Proportion of adult trees mortality by fire Minimum adult trees fire mortality rate = 0.038214 Maximum adult trees fire mortality rate = 0.04465	rate	Catry et al. (2012)

Table 1 (*Continued*)

| Variable		Description	Unit	Source
Cork oak population dynamics				
Random adult trees	death rate	Proportion of adult trees death	rate	Ribeiro and Surový (2008)
		Minimum adult trees death rate = 0		
		Maximum adult trees death rate = 0.000092		
DBH adult trees		Adult trees average diameter at breast height = 89	cm	Pulido et al. (2013)
Cork oak management income per year				
Random cork production per tree kg		Average kg of cork produced per tree	kg tree^{-1}	Pereira and Tomé (2004)
		Minimum cork production per tree = 35.95534		
		Maximum cork production per tree = 69.56406		
Random income from cork per tree euros kg		Average income in € per kg of produced cork	€ kg^{-1}	Pinheiro and Ribeiro (2013)
		Minimum income from cork per tree = 1.33		
		Maximum income from cork per tree = 3.33		
Average stripping cost cork extraction euros per kg		Average cost of cork extraction in € per kg	€ kg^{-1}	Pinheiro and Ribeiro (2013)
		Stripping cost cork extraction = 0.23		
Periodicity cork extraction		Average years between cork extraction	years	Pereira and Tomé (2004)
		Periodicity cork extraction = 9		
Average shrub clearing cost euros per ha		Average cost in € per cleared ha	€ ha^{-1}	Pinheiro et al. (2008)
		Shrub clearing cost = 120		
Average revenue from livestock euros per ha		Average income in € of livestock per ha	€ ha^{-1}	Pinheiro et al. (2008)
		Revenue form livestock = 22.4		
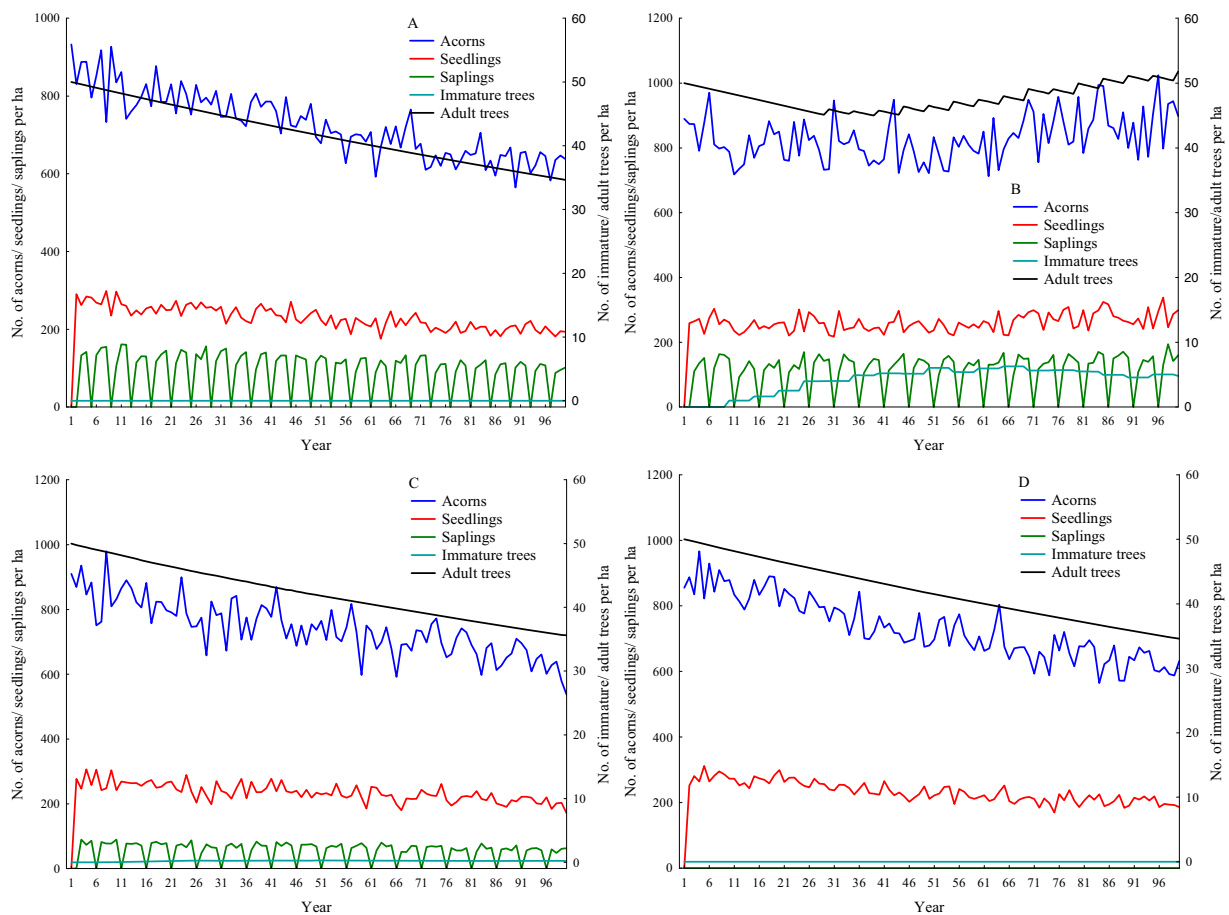
Fig. 3. Model simulations of cork oak population average trends per life stage (acorns, seedlings, saplings, immature, and adult trees) from 100 independent simulations throughout 100 years of simulation, including the stochastic occurrence of fires and droughts, as well as cork extraction every 9 years. (A) Scenario 1: shrub clearing every 4 years and presence of livestock (0.24–0.40 LU ha^{-1}). (B) Scenario 2: shrub clearing every 5 years and presence of livestock (0.24–0.40 LU ha^{-1}). (C) Scenario 3: shrub clearing every 5 years and the presence of livestock (0.40 LU ha^{-1}). (D) Scenario 4: shrub clearing every year, while livestock was absent.

3.1.3. Fire

Fires are common phenomena in southern Europe and cork oaks are vulnerable to its effects, especially seedlings and saplings, but also immature and adult trees when burning occurs immediately after cork extraction (Cabezudo et al., 1995; Pausas, 1997; Moreira et al., 2007; Catry et al., 2012). Data from the Spanish Forest Service (magrama.gob.es) was used to parameterize annual frequency and to model the consequences of fires for each cork oak life stage, taking into account the respective tree diameters (DBH) (Catry et al., 2012). The average number of fires (0.013 ± 0.017 fires.ha-

1; mean ± standard deviation) obtained per study unit in 44 years, from 100 independent stochastic simulations, was considered a reliable reproduction of the regional historical trends of fire events (0.016 ± 0.021 fires ha-1) (magrama.gob.es).

3.1.4. Drought

Droughts contribute to cork oak population losses as it hampers regeneration and increases tree mortality (Acácio and Holmgren, 2014). Considering the historical trends of droughts in mainland Portugal from 1976 to 2007 (INAG, 2007), the drought effects per year were considered as direct constrains over seedlings and saplings viability, the more vulnerable classes to this environmental factor. After the calibration procedures, the average number of 0.24 ± 0.078 droughts.ha^{-1} (mean ± standard deviation) obtained per study unit in 31 years, from 100 independent stochastic simulations, was considered a reliable reproduction of the regional historical trends of drought events (0.24 ± 0.43 droughts.ha-1) (INAG, 2007).

3.1.5. Livestock grazing

The consequences of intense grazing are the death of young trees, soil compaction, decrease of water infiltration and local biodiversity decline (Plieninger et al., 2004; Plieninger, 2006). Grazing pressure was defined as the ratio between stocking rate, *i.e.* the number of livestock units (LU) per unit area and unit time, and biomass (total dry weight of vegetation per unit area and unit time) (Allen et al., 2011; Sales-Baptista et al., 2015). From the 50's to present times, stocking rates in *montado* ecosystems increased from 0.10–0.15 to 0.24–0.40 LU ha^{-1} (Plieninger et al., 2004). Since the presence of livestock affects the survival of seedlings and saplings due to stem consumption, damages from livestock grazing were included in the model to assess the influence of livestock pressure in the cork oak regeneration. The livestock mortality rate was determined by the proportion of seedlings and saplings expected to die due to livestock grazing per year of simulation, ranging from 0.28 to 0.87 (Plieninger et al., 2004).

3.1.6. Shrub clearing

We included the effect of shrub clearing as a negative influence on seedlings and saplings viability since this is a common management practice in *montado* to control shrub invasion and to promote pasture growth (Pignatti, 1983; Pulido et al., 2001; Plieninger et al., 2003, 2004; Calvo et al., 2005). Contrary to the effect of shrub encroachment that enhances oak regeneration, promotes soil rehabilitation and prevents erosion, shrub clearing increases soil erosion and can derail the regenerating cork oak seedlings and saplings (Plieninger et al., 2004; Pulido and Díaz, 2005; Pérez-Ramos et al., 2008; Simões et al., 2009; Nunes et al., 2011; Arosa et al., 2015). Therefore, mortality due to shrub clearing was included in the model to assess the influence of this management practice in the cork oak regeneration.

3.1.7. Cork oak natural mortality

The cork oak natural mortality often represents a sum of continuous environmental and intrinsic processes with multiple contributors acting in the tree, such as age degeneration, insect damages or diseases, with unobvious cause-effect relationships concerning management practices and/or climate changes (Franklin et al., 1987; Branco et al., 2002; Camilo-Alves et al., 2013). In the model, we considered the effect of tree natural mortality over immature and adult cork oak trees using estimates from Ribeiro and Surový (2008) who determined the mortality trends of cork oak (trees ha^{-1}) in Portugal based on aerial photographs interpretation.

4. Economic balance

The cork oak *montado* is a multifunctional agro-forestry ecosystem that can benefit society with many goods and services. However, the economic sustainability of cork oak *montado* depends mainly on the price of cork, associated with its quality and final use. Cork oak trees may live 250–350 years but cork thickness and quality decreases and the limit to useful cork production is 150–200 years. Trees with circumference at breast height more than 70 cm can be debarked, which corresponds to trees with 25–40 years of age, depending on site productivity and tree density. After the first cork debarking, the minimum period tolerable between cork extractions is 9 years (Pereira and Tomé, 2004; Pinheiro et al., 2008). Furthermore, depending of the *montado* management and land uses profit, the benefits can be also increased mainly by livestock production. On the other hand, shrub clearing and cork extractions represent the main costs associated with the *montado* management. Overall, the modelling of cork oak population dynamics, under scenarios where different management practices were simulated, includes the assessment of the possible economic incomes from *montado* by balancing the financial gains and losses between scenarios.

5. Sensitivity analysis

According to Lee et al. (2015) the purpose of a sensitivity analysis is provide a measure of the robustness of the model, measuring the sensitivity of the obtained results to changes in parameters, forcing functions and/or sub-models. Local sensitivity analysis (SA) was done by one-parameter-at-a-time technique (OAT), changing the population parameters of the model with +/− 10% and +/− 50% variation of the respective values and observing changes in the response of the selected state variables (Ligmann-Zielinska, 2013).

6. Scenarios

Our model was constructed in order to simulate realistic environmental conditions and management practices concerning cork production, and assess its influence on the cork oak population dynamics. We designed the scenarios, considering the most common management practices in *montado*. The design included the more limiting factors for cork oak regeneration, the number of livestock units and the frequency of shrub clearing. Four scenarios were designed, considering fire and droughts as stochastic events and cork oak extraction every 9 years. In all scenarios a mean number of 50 adult trees.ha-1 was considered in the beginning of each simulation (t = 0), (Joffe et al., 1999), reproducing a typical *montado*. No seedlings, saplings and immature trees were considered in the beginning of simulations, since this is the typical situation in managed *montados*. Scenario 1 recreates the traditional management of the *montado*, with shrub clearing every 4 years and annual presence of livestock with values ranging between 0.24–0.40 LU ha^{-1} (the browsing index varied between 0.278–0.651 on seedlings and between 0.574–0.875 on saplings, Plieninger et al., 2004).

Scenario 2 was based on the scenario 1 trends, maintaining the density of livestock but with shrub clearing occurrence every 5 years. This allowed us to test the possible effects of a simple management alteration on the cork oak population dynamics.

In scenario 3, shrub clearing occurrence was set every 5 years as in scenario 2 but livestock density was increase (0.40 LU ha^{-1}). The browsing index corresponded to 0.651 and 0.875 on seedlings and saplings, respectively (Plieninger et al., 2004).

In order to recreate a *montado* with annual soil mobilization to produce fodder for livestock at the end of summer, the scenario 4 assumes shrub clearing every year and absence of livestock.

In all scenarios, mortality due to shrub clearing was based on the results of Arosa et al. (2015), assuming that all emerging seedlings and saplings die due to shrub clearing.

In order to detect possible effects of different management practices on the sustainability of the cork oak population, the Kruskal-Wallis non-parametric test was used since the variables tested did not meet the normality assumption, even after data transformation (McDonald, 2014): classes (i.e. average density of seedlings, saplings, immature trees and adult trees) and the income from cork oak management (i.e. average economic revenue), according to 2 time frames ($t = 50$ and $t = 100$) were compared between scenarios do detect long-term changes. In case of significant differences proved by the Kruskal-Wallis test, post hoc comparisons between the groups were done using mean ranks (Siegel and Castellan, 1988). Here, significance levels were corrected according to the sequential Bonferonni technique. Differences were considered significant if their probability of occurring by chance was less than 5%. Statistical analyses were carried out using the software Statistica version 8.0 (StatSoft Inc. 2007).

7. Results

7.1. Sensitivity analysis

The results from the OAT sensitivity analysis show that the parameters associated with seedlings, saplings, immature trees and adult tree cause individual changes on the modelling outcomes (Table 2). Random saplings cohort x livestock mortality rate was the parameter with the main key influence on the outputs of most surrogates.

7.2. Cork oak population dynamics

The scenario 1 involves the maintenance of *montado* (agro-forestry) traditional management: a decline of most classes (acorns, seedlings, saplings and adult trees) was predicted throughout the simulation period (Fig. 3A). Anyway the most limiting factor to the overall population is associated to the immature trees, constrained by mortality associated with shrub clearing. Fires, droughts, livestock browsing and cork extraction seem to play secondary roles. The decrease in the number of adult trees will also restrict the amount of acorns available each year, and consequently the number of viable seedlings and saplings (Fig. 3A): the initial density of 50 adult trees ha^{-1} is expected to decrease to circa 35 trees ha^{-1} in the end of simulation (i.e. −30% of adult trees/100 years).

Conversely, when the implementation of 1-year delay in shrub clearing was tested in scenario 2, the final density of adult trees is expected to sustain or even increase (52 trees.ha^{-1}) throughout (i.e. +4% of adult trees/100 years) (Fig. 3B). This slightly non-linear increase in the number of adult trees (Fig. 3), shows that some individuals reach the immature trees stage, compensating adults "natural mortality", raising the number of acorns and consequently the number of seedlings and saplings and creating a population of immature trees that will substitute the declining adults.

Nevertheless, the cork oak recovery achieved in scenario 2 by increasing shrub clearing periodicity for every 5 years, was completely withdrew by increasing livestock pressure as tested in scenario 3, with an overall decrease for all classes (Fig. 3C). The small amount of immature trees (Fig. 3C) is not enough to sustain in the long-term the population of adult trees. The estimated density of adult trees after 100 years of simulation decrease about 36 trees.ha^{-1} (i.e. −28% of adult trees/100 years).

Intensive shrub clearing effects modelled in scenario 4, resulted in neither saplings nor immature trees, being the most aggressive management as depicted in Fig. 3D. A decrease in the density of

adult trees to 35 trees ha^{-1} was estimated at the end of 100 years of simulation (i.e. −30% of adult trees/100 years).

By comparing the four scenarios we did not detect significant differences in the number of seedlings at $t = 50$ ($H_{3,96} = 5.95$, $p = 0.114$, Fig. 4A), while at $t = 100$ the number of seedlings in scenario 2 was significantly higher, with mean values of 299 ± 185.63 trees.ha^{-1} ($H_{3,96} = 58.56$, $p < 0.001$, Fig. 4A).

The low number of saplings in scenario 3, with mean values of 67.99 ± 64.05 at $t = 50$ and 63.08 ± 42.63 trees.ha^{-1} at $t = 100$, and the absence of saplings in scenario 4 at $t = 50$ and $t = 100$ was insufficient to withstand the population of immature and adult trees in the long-term ($H_{3,96} = 32.68$, $p < 0.001$ and $H_{3,96} = 29.06$, $p < 0.001$, for scenario 3 and 4 respectively. Fig. 4A).

The results for immature trees are similar at $t = 50$ and $t = 100$ ($H_{3,96} = 94.29$, $p < 0.001$, Fig. 4C). In fact, the obtained low number of immature trees in scenarios 1, 3 and 4 seems to be a consequence of the low availability of saplings due to intensive shrub clearing and presence of high number of livestock units. On the other hand, although immature trees have also been projected at low densities in scenario 2 (6.06 ± 5.90 and 4.81 ± 4.07 trees.ha^{-1}, at $t = 50$ and $t = 100$ respectively) it seems enough for the recovery of adult trees in the long term. This may be mainly due to the time to recover from the extended shrub clearing interval and the number of livestock units.

Regarding adult trees the results at $t = 50$ were significantly different between scenarios ($H_{3,96} = 46.65$, $p < 0.001$, Fig. 4D), being the scenario 2 the only one with increasing number of adult trees facing the other scenarios (46.53 ± 3.63 trees.ha-1). At $t = 100$, with 51.68 ± 8.19 trees.ha^{-1} in scenario 2 ($H_{3,96} = 60.77$, $p < 0.001$ m Fig. 4D) those differences were maintained

8. Montado management

The economic balances associated with scenario 1, 2, 3 and 4 resulted from the incomes related with cork extraction and livestock presence (and direct payments to farmers enrolled under the CAP) and losses from soil management and stripping cork costs. Interestingly, scenario 2 had the highest average predicted revenue from cork oak ($661.33 \pm 151.70 \, € \, ha^{-1}$ per year) ($H_{3,396} = 218.18$, $p < 0.001$, Fig. 5), while average profits associated with scenarios 1 and 3 were almost identical (564.94 ± 118.20 and 569.55 ± 120.70 € ha^{-1} per year, respectively). Scenario 4 generated the lowest average income ($275.13 \pm 112.39 \, € \, ha^{-1}$ per year).

9. Discussion

The novelty of our study was the inclusion of several work previously published within a single modelling platform with the key-components of changing *montado* agro-forestry, with holistic and ecological relevance, namely by focusing on drivers with influence on the long-term sustainability of these ecosystems. In the *montado*, a dramatic mortality of cork oak has been taking place when trees were thinned to ease the use of farm machinery and promote wider crowns that yield more acorns (Pulido et al., 2001; Plieninger, 2007). Some strategies were created to preserve the *montado* and enhance its cultural and natural values and ecological services. In this perspective, legal protection of the trees and agro environmental schemes have been implemented to protect and promote tree plantation in the *montado* (Pinto-Correia et al., 2011). On the other hand, most *montado* ecosystems in Spain and Portugal are private and changes in the rural socioeconomic condition, in part induced by the previous Common Agricultural Policy (CAP), have contributed to the rural exodus due to poor conditions for agriculture, the peripheral location of the rural areas and the lack of employment (Van Doorn and Bakker, 2007). There-

Table 2
Local sensitivity analysis (one-parameter-at-a-time) of the main state variables of the model to +/− 10% and +/−50% variation of the parameter values.

State Variable	Parameter	Sensitivity			
		−50%	−10%	+10%	+50%
Seedlings	Random seedlings shrub clearing mortality rate	0.046	−0.120	−0.308	−0.308
	Random seedling drought mortality rate	−0.184	−0.121	−0.036	[a]
	Random acorns germination rate	−0.597	0.045	−0.045	0.234
	Random acorns predation rate	−0.126	−0.004	−0.243	−0.174
	Random acorns insect damage rate	−0.136	−0.234	−0.177	−0.166
	Average seedlings fire mortality rate	−0.121	−0.121	−0.397	0.197
	Random seedlings livestock mortality rate	−0.010	0.190	0.155	−0.478
	Random saplings cohort × shrub clearing mortality rate	0.247	0.338	0.099	0.099
	Random saplings cohort × drought mortality rate	−0.184	−0.121	−0.036	[a]
	Average saplings fire mortality rate	−0.121	−0.121	−0.397	0.197
	Random saplings cohort × livestock mortality rate	1.228	2.245	−0.408	−0.377
	Random immature trees cohort × fire mortality rate	0.214	0.090	−0.102	−0.026
	Random immature trees cohort × death rate	−0.209	0.196	−0.099	0.025
	Period adult trees decay	−0.079	0.034	0.120	−0.203
	Adult trees per ha	−0.559	−0.333	0.420	−0.196
	Random adult trees death rate	−0.039	−0.051	−0.325	−0.189
	Random adult trees fire mortality rate	0.142	0.060	0.053	0.078
	Random acorns per tree	−0.663	−0.102	0.271	0.587
Saplings	Random seedlings shrub clearing mortality rate	1.658	1.599	1.225	1.225
	Random seedling drought mortality rate	2.595	1.193	1.360	[a]
	Random acorns germination rate	0.015	1.479	1.878	3.079
	Random acorns predation rate	1.538	2.054	1.405	0.257
	Random acorns insect damage rate	2.809	1.260	1.825	0.861
	Average seedlings fire mortality rate	1.656	1.411	2.118	2.076
	Random seedlings livestock mortality rate	4.098	1.384	1.335	−0.038
	Random saplings cohort × shrub clearing mortality rate	0.247	0.338	0.099	0.099
	Random saplings cohort × drought mortality rate	2.595	1.193	1.360	[a]
	Average saplings fire mortality rate	1.656	1.411	2.118	2.076
	Random saplings cohort × livestock mortality rate	11.535	7.816	0.502	0.286
	Random immature trees cohort × fire mortality rate	2.437	1.809	2.291	2.426
	Random immature trees cohort × death rate	1.643	1.729	1.277	2.287
	Period adult trees decay	1.629	1.019	2.932	1.979
	Adult trees per ha	0.252	2.158	4.240	3.693
	Random adult trees death rate	1.881	1.446	2.521	2.487
	Random adult trees fire mortality rate	1.384	1.421	0.765	2.113
	Random acorns per tree	−0.458	1.599	3.249	5.096
Immature trees	Random seedlings shrub clearing mortality rate	0.710	0.746	−0.011	−0.011
	Random seedling drought mortality rate	0.494	0.151	0.090	[a]
	Random acorns germination rate	−0.600	−0.020	0.367	1.614
	Random acorns predation rate	0.477	0.316	0.105	0.053
	Random acorns insect damage rate	0.479	0.548	0.056	−0.071
	Average seedlings fire mortality rate	0.156	0.434	0.078	0.191
	Random seedlings livestock mortality rate	1.123	0.362	0.137	−0.551
	Random saplings cohort × shrub clearing mortality rate	0.247	0.338	0.099	0.099
	Random saplings cohort × drought mortality rate	0.494	0.151	0.090	[a]
	Average saplings fire mortality rate	0.156	0.434	0.078	0.191
	Random saplings cohort × livestock mortality rate	5.273	3.918	−0.679	−0.991
	Random immature trees cohort × fire mortality rate	0.568	0.032	0.785	−0.061
	Random immature trees cohort × death rate	−0.294	0.779	0.427	0.602
	Period adult trees decay	−0.164	−0.207	0.033	0.483
	Adult trees per ha	−0.390	0.045	0.610	1.090
	Random adult trees death rate	0.356	−0.061	0.389	0.111
	Random adult trees fire mortality rate	1.003	0.354	0.293	0.177
	Random acorns per tree	−0.423	0.181	0.738	1.576
Adult trees	Random seedlings shrub clearing mortality rate	0.247	0.338	0.099	0.099
	Random seedling drought mortality rate	0.147	0.179	0.078	[a]
	Random acorns germination rate	−0.179	0.103	0.332	0.351
	Random acorns predation rate	0.178	0.249	0.086	0.001
	Random acorns insect damage rate	0.142	0.063	0.064	0.043
	Average seedlings fire mortality rate	0.233	0.179	0.246	0.140
	Random seedlings livestock mortality rate	0.454	0.156	0.307	−0.160
	Random saplings cohort × shrub clearing mortality rate	0.247	0.338	0.099	0.099
	Random saplings cohort × drought mortality rate	0.147	0.179	0.078	[a]
	Average saplings fire mortality rate	0.233	0.179	0.246	0.140
	Random saplings cohort × livestock mortality rate	2.016	2.006	−0.205	−0.381
	Random immature trees cohort × fire mortality rate	0.160	0.165	0.223	0.223
	Random immature trees cohort × death rate	0.154	0.245	0.135	0.229
	Period adult trees decay	−0.098	0.109	0.215	0.419
	Adult trees per ha	−0.421	0.138	0.504	0.795
	Random adult trees death rate	0.248	0.086	0.251	0.255
	Random adult trees fire mortality rate	0.227	0.228	0.252	0.155
	Random acorns per tree	−0.150	0.175	0.203	0.572

[a] Sensitivity was not calculated considering that with the implemented variation, the parameter attained values higher than 100%.
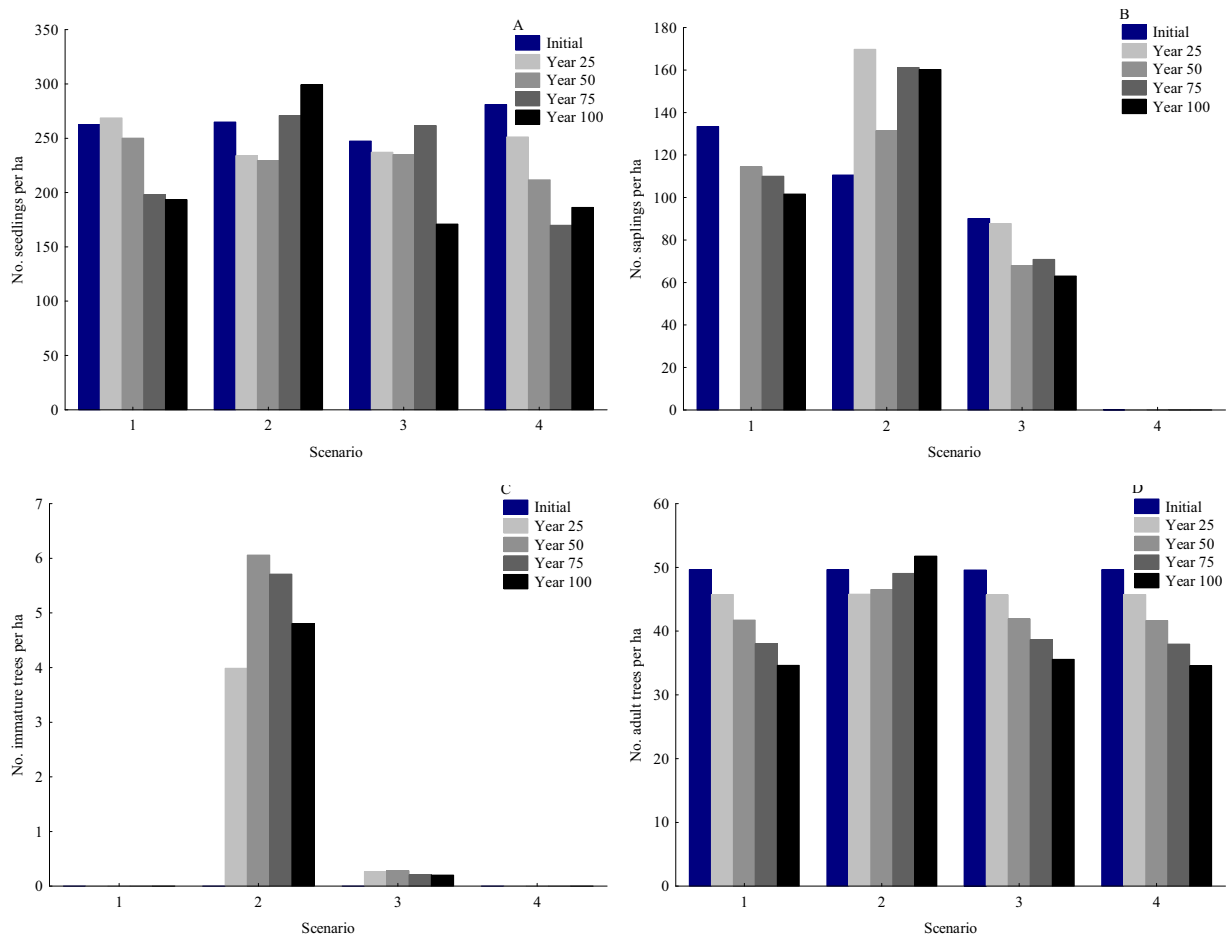
Fig. 4. Average density of seedlings (A), saplings (B), immature trees (C) and adult trees (D) per hectare, from 100 independent simulations, according to 4 time frames ($t = 1$, $t = 25$, $t = 75$ and $t = 100$) in each scenario considered.

fore, the importance of the socioeconomic situation in relation to land use change is generally recognized (Brandt et al., 1999): the direct aid to cereal areas and to some grazing animals into single payments to farmers, which are decoupled from production, resulted in land owners that have tended to replace sheep and goats with cattle, which in last 16 years increased its numbers by 2.5 times due to CAP support (Dýrmundsson, 2004; Pinto-Correia and Godinho, 2013; Pinto-Correia et al., 2014; Viegas et al., 2014). After the abandonment of cereal crops and livestock rearing, during the 70's, shrub clearing and rotational ploughing became common practices to control shrub invasion, to promote pasture production and prevent fire (Pignatti, 1983; Pulido et al., 2001; Plieninger et al., 2003, 2004, 2005). The heavy cattle breeds or higher livestock rates, together with deep ploughing of cultivated crops, is undoubtedly affecting natural regeneration and many authors estimate these factors as a major threat to the *montado* ecosystem (Pinto-Correia and Mascarenhas, 1999; Bugalho et al., 2011; Moreno and Pulido, 2009).

10. Main limitations to regeneration

Overall, our results indicate that the main limitations to the lack of cork oak regeneration in *montado* ecosystems rise from land management practices, rather than environmental factors. In fact, the obtained results suggest that delaying shrub clearing and limiting the quantity of livestock units might constitute a suitable measure to allow the long-term sustainability of the *montado*. A minimal delay of one year in soil ploughing will be enough for

landowners to ensure the maintenance/increase in the number of immature and adult trees as well as the long-term sustainability of the *montado* ecosystem. Since cork is the most profitable product obtained from the *montado*, an increase in the density of adult trees would have also a direct positive effect on the economic balance of agroforestry "crop". On the other hand, by increasing livestock densities, cork oak regeneration is severely constrained due to the consumption of seedlings and saplings, and the regeneration cycle would eventually collapse. A similar result is obtained when there is a continuous use of heavy machinery. Other conventional semi quantitative studies that investigated the effects of *montado* changes have come to similar conclusions, corroborating that livestock pressure acts as a main driver determining the rates of *montado* loss, although not explaining with detail the process beneath (Blondel, 2006; Plieninger, 2007; Gaspar et al., 2008; Bugalho et al., 2011). Acorn predation and browsing, as well as trampling of seedlings are the main pressures of overgrazing, associated with the intensity and type of grazing (livestock type, breeds, density, length of time in pasture) (Pulido and Díaz, 2005). Our results identify limits to the carrying capacity of the *montado* near 0.40 LU ha-1, from which the future of the *montado* sustainability would be seriously threatened. This outcome also demonstrates that care must be taken with the CAP support to increasing the number of livestock per ha. Also Baeza, 2004 and Calvo et al., 2012 limit the number of LU.ha-1 near 1, which based on our results, may put the *montado* agroecosystem at risk.

Results relative to shrub clearing demonstrated that most of the *montado* loss is associated with land management practices.
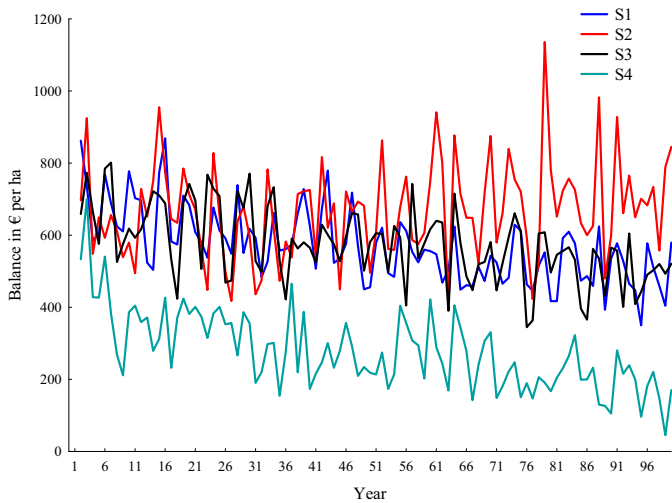
Fig. 5. Predicted average economic balance (€.ha-1) for scenarios 1, 2, 3 and 4, from 100 independent simulations throughout 100 years of simulation.

Shrub clearing might hinder landscape heterogeneity and damage the fungal community in *montado*, decreasing the mycorrhizal community that is part of the symbiosis most intimately connected to the soil and most directly involved in uptaking nutrients and influencing soil properties (Azul, 2002; Azul et al., 2010). The period between consecutive shrub clearings supports different management objectives: intensive shrub clearing every three years will favour pastureland for livestock grazing and decreasing shrub growing will neglect its positive protector effect over regeneration (Canteiro et al., 2011; Simões et al., 2015). This agrees with our results, where short periods between clearings would inhibit the regeneration of trees, since mechanical clearing is a non-selective tool that destroys most seedlings (Arosa et al., 2015), thus the absence of immature trees directly affects negatively the population of adult trees. According to our long term results, we assume that extending shrub clearing for longer periods would benefit natural regeneration, which is in agreement with other short term studies (Canteiro et al., 2011; Godinho et al., 2014; Simões et al., 2016). In fact, other authors' conclusions suggest maintaining shrub cover for a period between 7 and 12 years to help cork oak regeneration and promote tree nursery and soil rehabilitation. Conversely, periods longer than 12 years could facilitate fire spread due to the accumulation of dead material (Canteiro et al., 2011). EU programs have only been subsidizing planting young cork oak trees but results obtained by Arosa et al. (2015) confirmed that direct seeding (or natural regeneration) of cork oak is a good alternative to facilitate regeneration, because emergence success ranged between 43.5% and 63.9%. Agricultural policies should consider this information as they play an important role in management decision and can contribute to further degradation of tree cover (Pinto-Correia and Vos, 2004).

11. Future perspectives

The developed dynamic model allowed predicting long term trends in cork oak local population, including those attributable to changes in habitat due to different sources of disturbance and management scenarios (Winkler, 2006). Therefore, this study represents an early step to support strategic options for impact mitigation and management by providing projections of long-term indicator trends under realistic social–ecological change scenarios (Bastos et al., 2015). Although the projections recreates realistically the known cork oak regeneration patters occurring in *montado*

regions, some limitations arise when considering validation as a fundamental process to assess the relative accuracy of the model response facing independent real data (Rykiel, 1996). The validation of the predictions was not possible in this study due to the impracticality of an immediate validation of future scenarios, which can only be done after several years of collecting relevant site-specific information under well-known local environmental conditions (Glenz et al., 2001; Chaloupka, 2002).

Our academic and user-friendly model, after specific calibration and validation, could be used to support specific farmer decisions in the management of cork oak savannas and support agroforestry extension services. Furthermore, large-scale analyses are mandatory to fully understand the role of multi-scale cumulative drivers influencing the regeneration process and the role of policies in managing this ecosystem (Godinho et al., 2014). In this context, the combination of local population dynamics (temporal) and regional distributional (spatial) models can help to assess and predict how anthropogenic and environmental changes will affect the abundance and displacement of vulnerable species or communities in disturbed ecosystems and regions (Guisan and Thuiller, 2005). Besides, models are useful to identify areas where the conflict between the previous forecasted trends and drivers of pressure are of major conservationist concern, by allowing the integration of the ecological consequences from local to regional levels (Bjørnstad et al., 1999; Vicente et al., 2011). Model-based research resulted very useful as an investigative tool to predict the outcome of alternative scenarios, guiding current restoration options from predicted future targets (Bastos et al., 2012). Future work should be directed to develop the restoration strategies of cork oak *montados*, which will contribute to fight desertification, protect biodiversity, mitigate climate change and promote the sustainable development of local communities.

12. Conclusion

The capacity to accurately predict responses of species to landscape changes is crucial for conservation planning and to support key ecosystems management (Kandziora et al., 2013). In *montado* ecosystems, there is a lack of evaluation on the long-term ecological changes associated with management practices, whose consequences might represent major limiting factors to the ecological and economic sustainability of this valuable agro-ecosystem. The developed dynamic model allowed integrating and simulating the main factors affecting the long-term sustainability of the *montado* system, namely the major limiting factors driving the lack of regeneration of cork oak trees in *montados*. The number of livestock units per ha and short shrub clearing intervals were identified as the main limiting factors for the long term sustainability of this agroforestry ecosystem. Thus, it should be considered a maximum quantity of livestock of 0.40 LU.ha^{-1} and the minimum periodicity between soil ploughing to 5 years. Specific management practices are suggested to overcome the sustainability of the identified problems.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecolmodel.2016.10.008.

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