



Quercus suber forest and *Pinus* plantations show different post-fire resilience in Mediterranean north-western Africa

Brahim Chergui¹ · Soumia Fahd¹ · Xavier Santos²

Received: 1 December 2017 / Accepted: 16 May 2018 / Published online: 11 June 2018
© INRA and Springer-Verlag France SAS, part of Springer Nature 2018

Abstract

• **Key message** In the African rim of the Western Mediterranean Basin, cork oak forests and pine plantations coexist. Under similar fire regimes, cork oak forest is more resilient in terms of habitat structure (canopy, understory, and complexity of vegetation strata) than pine plantation. By contrast, both woodland types show similar resilience in plant species composition. Resilience in habitat structure varies between the two woodland types because of the resprouting and seeding strategies of cork oak and pine species, respectively. These differences can be relevant for the conservation of biodiversity of forested ecosystems in a future scenario of increased fire frequency and scale in the Mediterranean basin.

• **Context** Wildfires have major impacts on ecosystems globally. In fire-prone regions, plant species have developed adaptive traits (resprouting and seeding) to survive and persist due to long evolutionary coexistence with fire. In the African rim of the Western Mediterranean Basin, cork oak forest and pine plantation are the most frequently burnt woodlands. Both species have different strategies to respond fire: cork oak is a resprouter while pines are mostly seeders.

• **Aims** We have examined the hypothesis that pine plantations are less resilient in habitat structure (canopy, understory, diversity of vegetation strata) and plant composition than cork oak woodlands.

• **Methods** The habitat structure and plant species composition were measured in 30 burnt and 30 unburnt 700-m transects at 12 burnt sites from north-western Africa, where the two forest types can coexist. Habitat structure and plant species composition were compared between burnt and unburnt transects from cork oak and pine plantation woodlands with generalized linear mixed models and general linear models.

• **Results** The results showed significant interaction effect of fire and forest type, since cork oak forest was more resilient to fire than was pine plantation in habitat structure. By contrast, both forest types were resilient to fire in the composition of the plant communities, i.e., plant composition prior to fire did not change afterwards.

• **Conclusion** The higher structural resilience of cork oak forest compared to pine plantation is related to the resprouting and seeding strategies, respectively, of the dominant tree species. Differences in the responses to fire need to be considered in conservation planning for the maintenance of the Mediterranean biodiversity in a future scenario of changes in fire regime.

Keywords Cork oak forest · Pine plantation · Fire · Habitat structure · Resilience · Rif · Mediterranean basin

Handling Editor: Paulo M. Fernandes

Contribution of the co-authors B.C., X.S., and S.F. conceived the idea; B.C. collected the data in the field; B.C. analyzed the data; and all the authors contributed to the writing.

Electronic supplementary material The online version of this article (<https://doi.org/10.1007/s13595-018-0742-6>) contains supplementary material, which is available to authorized users.

✉ Brahim Chergui
brahim_chergui@ymail.com

² CIBIO/InBIO, Centro de Investigação em Biodiversidade e Recursos Genéticos da Universidade do Porto, R. Padre Armando Quintas, 4485-661 Vairão, Portugal

¹ Equipe de Recherche Ecologie, Systématique, Conservation de la Biodiversité, Faculté des Sciences de Tétouan, Université Abdelmalek Essaâdi, BP 2121 El M'Hannech, Tétouan, Morocco

1 Introduction

Fire is a natural process in many ecosystems and a key element to understand their functioning and plant community composition (Moritz et al. 2012; Bond et al. 2005). In fire-prone regions, the effects of fire on vegetation are usually the most obvious impacts of burning (DeBano et al. 1998), and this variation may influence biodiversity at local and landscape scales (De Grandpré et al. 2000; Burrows 2008). Moreover, the long history of fire has strongly selected against fire-sensitive plant species and has promoted a wide range of fire-resistant strategies (Pausas et al. 2008). Thus, forest ecosystems can recover their original structure and composition following a fire due to the resilience of plant community (Trabaud and Lepart 1980; Keeley 1986; Retana et al. 2002). Post-fire resilience is based on the ability of plant species to generate new shoots from dormant buds located on fire-resistant structures after stems have been fully scorched (hereafter resprouters) and/or to generate a fire-resistant seed bank stored in the soil or in the canopy with seeds that germinate profusely after blazes (hereafter seeders; Keeley 1986; Hodgkinson 1998; Pausas et al. 1999; Pausas 2001).

A forest ecosystem can be considered resilient if it is able to recover its composition, structure, and main functions following a disturbance (Folke 2006). Forest resilience can be partitioned in two components: (i) structural resilience when the habitat after fire recovers soon in terms of canopy, understory, and complexity of vegetation strata and (ii) compositional resilience when the plant communities after fire resemble those of pre-fire conditions in terms of species composition and relative abundance (Drever et al. 2006; Lipoma et al. 2016). In fire-prone regions, forest resilience can be disrupted by changes in fire regimes (greater fire frequency and extent) because of socioeconomic changes (e.g., land abandonment and fuel increase; Moreira et al. 2001; Moreira and Russo 2007; Pausas and Fernández-Muñoz 2012; Pausas et al. 2012). Likewise, the introduction of very flammable species in monocultures such as coniferous trees also can disrupt forest resilience (Shakesby et al. 1996; Pausas et al. 2004; Vallejo et al. 2006). This process is occurring in Western Mediterranean landscapes as coniferous species (seeders) are replacing oak species (resprouters) (Aronson et al. 2009; Costa et al. 2011). Seeder species recover after fire from seeds stored in the soil or in the canopy (Pausas and Keeley 2014). By contrast, resprouters often have lower seed recruitment than seeder species (Keeley 1986; Burgman and Lamont 1992; Pausas 2001), although they resprout after fire from basal lignotuber or epicormic stem buds (Clarke et al. 2013). The establishment of seedlings requires more time and thus their regeneration is slower than resprouting. These opposing responses of fire make resprouter species faster fire resilient than seeders (Lloret 1998; Pausas 1999; Rodrigo et al. 2004; Valdecantos et al. 2009).

The forest cover in the African rim of the Western Mediterranean Basin accounts for 35% of the total area. The commonest native tree species is the cork oak *Quercus suber*. In recent decades, however, cleared or degraded *Q. suber* forest and shrublands have been replaced by coniferous plantations composed mostly of maritime pine *Pinus pinaster* (Pastor-López et al. 1997). Wood and resin from pine plantations are important socioeconomic resources which are reflected in the intensity of reforestation. These reforestations have been undertaken often in the bioclimatic subhumid and humid zones of the Rif and eastern Middle Atlas (Emberger 1955), representing about 90% of all reforestation in the region (Belghazi and Romane 1994).

The cork oak resprouts after fire from basal lignotuber but to a greater extent from epicormic stem buds (Molinas and Verdaguer 1993). This species is adapted to the impact of fire due the capacity of the bark to protect against high temperatures (Barberis et al. 2003; Úbeda et al. 2006). By contrast, maritime pine and other *Pinus* species are susceptible to crown fires, particularly at the juvenile stage (Cruz and Fernandes 2008; Molina et al. 2011; Mharzi Alaoui et al. 2017). Pines regenerate from seed (Díaz-Delgado et al. 2002) especially in serotinous populations (Hernández-Serrano et al. 2013). Although *P. pinaster* and other pines can survive low-intensity surface fires (Cтры et al. 2010; Vega et al. 2010; Keeley 2012), it is considered low-resilience species (Proença et al. 2010). This pine species is an obligate seeder with low but variable degree of serotiny depending on populations (Tapias et al. 2004; Hernández-Serrano et al. 2013). Moreover, *P. pinaster* seedlings are shade intolerant and recruitment can be highly heterogeneous and influenced by a combination of climatic, edaphic, and structural factors (Rodríguez-García et al. 2011). Therefore, post-fire regeneration of *P. pinaster* forest can be slow, and also pines can be replaced by more competitive species such as *Quercus* trees in early post-fire stages (Torres et al. 2016).

In this study, we compare the structural and compositional resilience of cork oak forests and coniferous plantation in an area located on the African rim of the Western Mediterranean Basin. This area is affected by Mediterranean fire regime (crown and summer fires; Chergui et al. 2017), and *Q. suber* forest and *P. pinaster* plantations are the commonest burnt woodlands. Due to the opposing responses to fire on the part of *Q. suber* and *P. pinaster* (resprouter and seeder, respectively), we expect marked differences in post-fire structural resilience between these two forest types, as well as differences in compositional resilience. *Quercus suber* recovers very quickly because it resprouts directly from the trunk and branches and this can allow its re-establishment in a short time period (Pausas 1997; Carrión et al. 2000; Alanís-Rodríguez et al. 2011). For this reason, we hypothesize that *Q. suber* forests will present higher structural and compositional resilience following fire than *P. pinaster* plantations.

2 Material and methods

2.1 Study area and fire regime

The study area is located in north-western Africa between 35° 00' and 35° 55' N and between 5° 00' and 6° 15' W, covering nearly 12,650 km² (Fig. 1). The maximum elevation is 2159 m a.s.l. at Jbel Lakraa. The climate is mainly of the Mediterranean type with an altitudinal gradient in temperature and precipitation from valleys and coast to uplands. Mean annual temperatures vary from 15 to 19 °C (Ajbilou 2001; Ghallab and Taiqui 2015), whereas the annual rainfall in some areas of the Rif reaches 2000 mm.

Vegetated areas cover 396,165 ha, which represents 35% of the total area of the region including natural woodlands and scrublands (84.5%) as well as plantations (15.5%; Benabid 2007). The forest coverage is dominated by cork oak *Quercus suber* and scrubland formations, followed distantly by Holm oak *Quercus rotundifolia* and maritime pine *Pinus pinaster*. In the study area, natural *P. pinaster* and *Pinus halepensis* populations are distributed discontinuously, covering small, patchy areas (Quezel et al. 1992; Belghazi et al. 2000; Benabid 2007). Plantations include 49,124 ha of coniferous (mainly *P. pinaster*) and 14,462 ha of deciduous species (mainly *Eucalyptus*) (Mharzi Alaoui et al. 2015). The study area undergoes a

typical Mediterranean fire regime with blazes occurring in the hot, dry summer season (Chergui et al. 2017). Two major forests are affected by fire, i.e., native cork oak forests and pine plantations, in which fires are likely medium- or small-size fires (< 1000 ha) due to the intense fuel (wood) use linked to the particular socioeconomic conditions of northern Morocco (Chergui et al. 2017).

2.2 Site selection and vegetation sampling

The vegetation was sampled at 12 field sites (Fig. 1; Table S1) in the spring of 2016. These sites were selected according to the fire extent and year of fire occurrence between 2008 and 2015 (8 years to 1 year = time since fire; TSF thereafter). The fire history at each site was taken from the Haut Commissariat aux Eaux et Forêts et à la Lutte Contre la Désertification (hereafter the HCEFLCD) and from Mharzi Alaoui et al. (2015). Most of the fires in both forest types occurred in summer (Table S1) as is characteristic of the Mediterranean region in general (Chergui et al. 2017). The forest at the study sites is mostly composed by *Q. suber* and *Pinus pinaster* plantations. Some burnt plantations were composed by other *Pinus* species such as *P. halepensis*, *Pinus brutia*, and *Pinus pinea* (Table S2; Mharzi Alaoui et al. 2017). These coniferous species have different degree of serotiny, but all of them are obligate seeders (Thanos and Doussi 2000; Pausas et al. 2004; 2008);

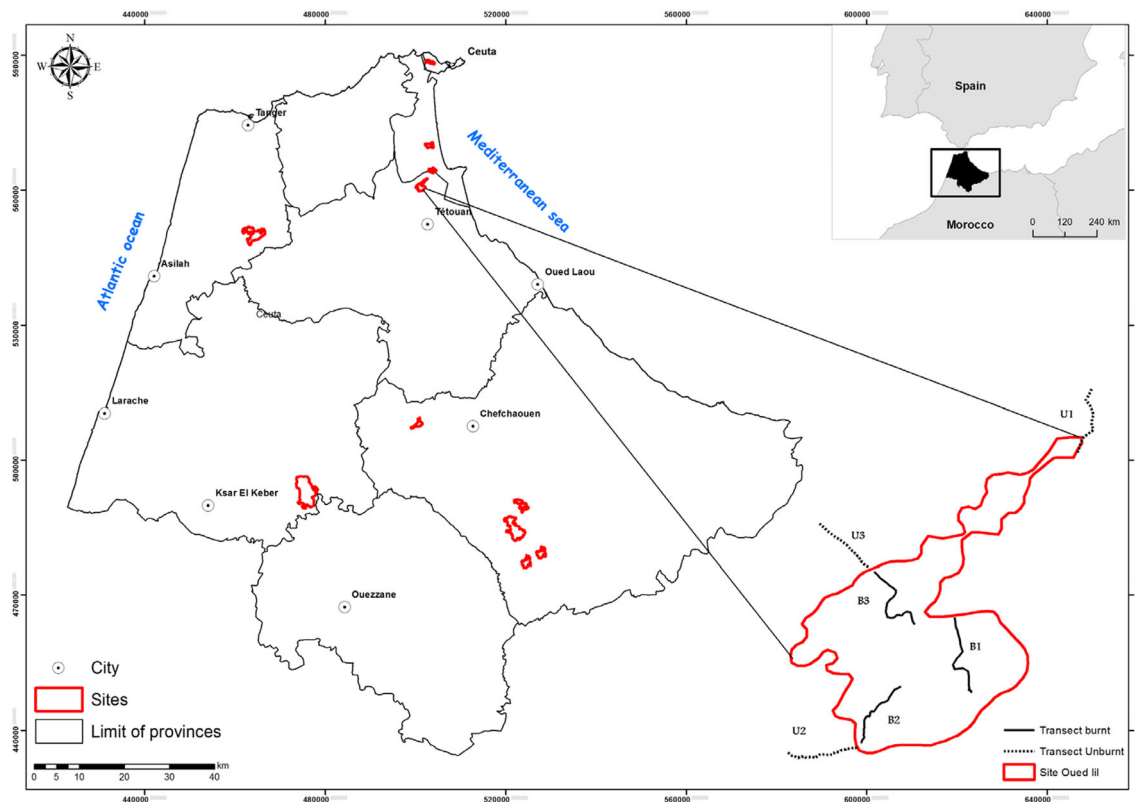


Fig. 1 Location map at the African rim of the Western Mediterranean Basin, fire sites (red polygons), and example of the spatial scheme of three burnt and three unburnt transects from site Oued Lil

therefore, all burnt pine transects were pooled as pine plantation sites for further analyses.

The 12 sites covered an altitudinal gradient from 97 to 1281 m a.s.l. Average rainfall and annual temperature at each site (range 16.20–19.43 °C and 631–965 mm respectively; Table S1) were obtained from Climatology Resource for Agroclimatology (<https://power.larc.nasa.gov/cgi-bin/agro.cgi?na>). Average rainfall and annual temperature were positively correlated with elevation ($r = 0.547$, $P = 0.002$ and $r = 0.469$, $P = 0.005$, respectively). We found no differences in the distribution of the two forest types (*Quercus* and *Pinus*) with respect to elevation (Mann-Whitney U test, $Z = -1.90$, $P = 0.06$). Thus, we discarded biases in the resilience of cork oak and pine plantation burnt sites due to climate (e.g., rainfall) effects (Walker et al. 2016).

At each site, burnt polygons were laid out from Landsat imagery provided by the USGS Earth Explorer server (<http://earthexplorer.usgs.gov/>), and 60 transects (2–3 burnt and 1–3 unburnt at each site) were randomly assigned for vegetation sampling. Unburnt (control) transects were selected around each burnt area and usually near each unburnt pair (the maximum distance between the burnt-unburnt pair was 2 km). We assumed that vegetation in control areas was left unchanged and reflected initial vegetation composition before the fire occurred. Transects followed small unpaved trails were located at least 50 m from the burnt edge left by the fire and averaged $698 \text{ m} \pm 0.93$ length. The elevation and geographic location of each transect were recorded with a Global Positioning System (GPS).

2.3 Variable selection and data analysis

At each transect, habitat structure and species composition were measured on 50 quadrats (25 quadrats on each side of the transect). Quadrats were $10 \text{ m} \times 10 \text{ m}$ and located 15 m apart. In each quadrat, nine habitat structure and four species composition variables were recorded (Table 1). Structural variables were arranged roughly into three vegetation layers: the

overstory, defined as trees, which were dominant in the uppermost canopy; the midstory, composed mainly by shrubs; and the understory, composed of grasses and other ground covers. Thus, structural variables were defined as the extent (percentage of cover) of three vegetation types, i.e., tree, shrub, and grass, as well as three ground cover types, i.e., bare ground, litter, and rocks. Percentage of cover for structural variables was visually estimated at each quadrat. Additionally, we estimated the maximum tree height at each quadrat and calculated the canopy using a convex spherical densiometer. At each quadrat center, the densiometer was mounted on a tripod oriented in direction of the four cardinal points and four canopy measurements were taken. The canopy closure was the average of the four cardinal point measurements (Lemmon 1957). For the structural variables (tree, shrub, grass, bare ground, litter, rock cover, maximum tree height, and canopy), we calculated average values at each transect using individual values of each quadrat. Additionally, we estimated the structural complexity of each transect by calculating the Simpson diversity index from the percentage of all vegetation and ground cover types in each quadrat. This index equals the diversity of vegetation and ground layers; therefore, higher Simpson diversity scores indicate more complex (heterogeneous) transects.

Compositional variables were based on the cover of all perennial and the commonest herbaceous plant species ($n = 43$) identified at each quadrat. Plant species cover is a methodological surrogate of vegetation composition and abundance (Daubenmire 1959). The cover of each species followed Daubenmire's (1959) procedure: at each $10 \times 10 \text{ m}$ quadrat, each species cover was visually estimated using a cover classification as follows 1, 0.5%; 2, 3.5%; 3, 15.5%; 4, 38.0%; 5, 63.0%; and 6: 88.0%. For each plant species identified along the transect, we calculated average values using cover scores of all quadrats. Average values of each plant species found at each transect were used to calculate the Simpson diversity index and to run a principal components analysis (PCA) to reduce all the variability of transects in terms of plant composition to a low number of

Table 1 List and description of the structural and compositional vegetation variables (nine and four, respectively) examined in burnt and unburnt transects at 12 sites located in north-western Africa

Variable	Description
Canopy	Measured using a convex spherical densiometer in each quadrat
Tree height	Average height (m) of the tallest tree estimated from 100 $10 \times 10 \text{ m}$ quadrats per transect
Tree ^a , shrubs, grass, bare ground, litter, and rock cover	Average percentage cover estimated from 100 $10 \times 10 \text{ m}$ quadrats per transect
Simpson diversity (STR)	Calculated from the percentage of all vegetation and ground cover types
Number of plants ^a	Recorded at each transect
PC1 and PC2	Axes of the principal component analysis based on the cover of each plant species estimated at each transect
Simpson index (SP)	Calculated from the cover of each plant species estimated at each transect

^a The excluded variables to perform statistical models due to the high correlation ($r > 0.7$)

variables. Thus, the total number of plant species found in each transect (all perennial and the commonest herbaceous plant species), the Simpson diversity index, and the first two axes of the PCA (PCA1 and PCA2) were used as compositional variables in further analyses (Table 1).

From the 13 variables initially considered, we removed tree cover and the total number of plants (perennial and the commonest herbaceous) as they were highly correlated ($r > 0.7$) with canopy and PCA1-PCA2 respectively. The 11 variables retained were used as dependent variables in further multivariate models. We used linear models to examine the effect of elevation, fire (burnt vs. unburnt), forest type (*Q. suber* forest and *P. pinaster* plantation) on eight structural and three compositional variables. Generalized linear mixed models (GLMMs) with site as a random effect were conducted for seven dependent variables (canopy, tree height, shrub cover, grass cover, bare ground cover, litter cover, and rock cover) with a Poisson distribution as the data are non-negative values and counts, e.g., cover classes and tree height estimation (Table 2). General linear models (GLMs) were conducted for four dependent variables (Simpson index [STR], Simpson index (SP), PC1 and PC2) with Gaussian distribution (Table 2). GLMs and GLMMs were performed using the lme4 package (Bates et al. 2015) and loess smooth curves using the ggplot2 package (Wickham 2009). All the statistical analyses were made with R (R Development Core Team 2015).

Data availability The datasets generated during and/or analyzed during the current study are available from the corresponding author on reasonable request.

3 Results

3.1 Structural resilience

We detected significant variation in almost all structural variables (canopy, tree height, shrub cover, bare ground cover, litter cover, rock cover, and Simpson index) between burnt and unburnt transects (Table 2; [supplementary material data set](#)). Significant differences also appeared between the two forest types (cork oak and pine plantation) for canopy, rock cover, and Simpson index (Table 2). Notably, we found significant differences in the interaction fire \times forest type for canopy, tree height, shrub cover, and bare ground cover (Table 2). Shrub cover and the Simpson index remained similar between burnt and unburnt cork oak transects, whereas strong variation arose in pine plantations. In burnt pine plantations, fire significantly decreased canopy, tree height, and shrub cover, whereas bare ground cover increased (Fig. 2). A forest type effect on the habitat-complexity resilience was observed: the Simpson index of the cork oak forest did not change significantly with fire, whereas it diminished in burnt

pine plantations (Fig. 2g). We found no significant differences of structural variables with the elevation of the transects except rock cover (greater at higher elevations).

3.2 Compositional resilience

A total of 45 and 45 plant species were recorded in burnt and unburnt transects, respectively, belonging to 23 taxonomic families (Table S3). We found no differences in the diversity of plant communities between the two forest types (Table 2). However, plant species composition differed between cork oak and pine plantation transects according to the values of the first and second PCA axes (PC1: eigenvalue 5.46, 12.7% of explained variance; PC2: eigenvalue 5.10, 11.9% of total variance explained; Table 2, Fig. 3). Among the commonest plant species, those that best explained the shift in plant composition between cork oak forest and pine plantation were *Cistus monspeliensis* and *Pistacia lentiscus*, which were more abundant in pine plantations, whereas *Arbutus unedo*, *Cistus salvifolius*, and *Cistus crispus* were more abundant in cork oak forest (Table S4). Plant species composition (PC1 and PC2 scores) also varied with elevation.

4 Discussion

4.1 Vegetation structure

The most evident result of our study was that fire simplifies the complexity of the forest regardless of the type of forest studied; this trend was especially visible in canopy and tree height (overstory), which decreased after burning. Notably, several structural variables showed significant differences in the interaction between fire and forest type variables. These results did support the hypothesis that cork oak forest had greater structural resilience following fire disturbance than did coniferous plantations. These differences between forest types suggest that less diversified systems such as pine plantations are less resilient to natural disturbance than are more complex ecosystems such as native *Q. suber* forest (Drever et al. 2006). Besides differences between native forest and pine plantations, variation in local environmental and weather conditions can also play a role in local resilience to fire (Proença et al. 2010; Walker et al. 2016). However, we detected no effect of elevation (a surrogate of temperature and rainfall) on structural resilience, suggesting that variation of local environmental gradients was less important than forest characteristics to explain resilience to fire.

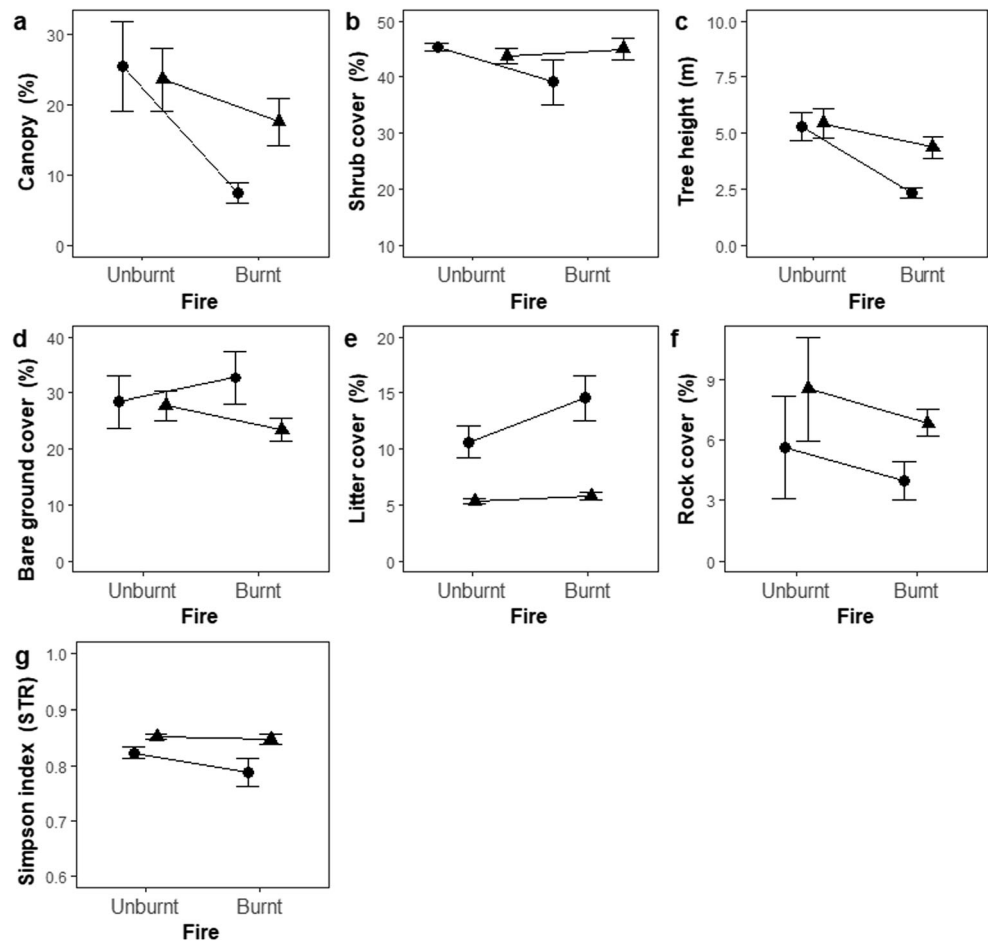
Although large (= old) cork oak trees can be eliminated by fire due to their reduced ability to resprout, canopy and tree height (overstory variables) did not significantly vary after fire in the cork oak forest from the study area. By contrast, we observed a great decline in canopy and tree height from pine

Table 2 Summary results of the generalized linear mixed models (GLMMs) and general linear model (GLMs) analyzing the influence of fire on structure (STR) and composition (SP) of the vegetation in burnt sites from north-western Africa

Response variable	Model type	Fire			Forest types			Fire*forest			Elevation		
		Estimate	<i>t/z</i> value	<i>p</i>	Estimate	<i>t/z</i> value	<i>p</i>	Estimate	<i>t/z</i> value	<i>p</i>	Estimate	<i>t/z</i> value	<i>p</i>
Canopy	GLMMs	1.225	11.403	< 0.0001	0.702	3.591	0.0003	-0.962	-7.093	< 0.0001	-0.0001	-0.597	ns
Tree height	GLMMs	0.838	4.128	< 0.0001	0.452	1.295	ns	-0.643	-2.436	0.014	-0.0004	-0.969	ns
Shrub cover	GLMMs	0.158	2.817	0.0048	0.115	1.197	ns	-0.192	-2.422	0.015	-0.00010	-0.518	ns
Grass cover	GLMMs	-0.023	-0.342	ns	-0.004	-0.035	ns	0.129	1.322	ns	0.00011	0.486	ns
Bare ground cover	GLMMs	-0.183	-2.786	0.0053	-0.112	-1.058	ns	0.336	3.389	0.0007	0.00010	0.440	ns
Litter cover	GLMMs	-0.258	-2.488	0.0129	-0.212	-0.903	ns	0.170	0.898	ns	0.0000	0.151	ns
Rock cover	GLMMs	0.356	2.118	0.0342	0.503	2.518	0.0118	-0.149	-0.691	ns	0.0012	3.224	0.0012
Simpson index (STR)	GLMs	0.0000	2.126	0.0378	0.0000	2.474	0.0165	0.0000	-1.275	ns	0.0000	1.274	ns
PC1	GLMs	-0.195	-0.275	ns	2.064	2.854	0.0060	0.889	0.869	ns	0.0031	5.498	< 0.0001
PC2	GLMs	0.028	0.044	ns	3.550	5.107	< 0.0001	-0.202	-0.214	ns	-0.003	-4.262	< 0.0001
Simpson index (SP)	GLMs	0.0000	0.127	ns	0.0000	0.225	ns	0.0000	-0.086	ns	0.0000	-0.054	ns

z values are provided for data modeled with Poisson distribution and *t* values for those modeled with Gaussian distribution. Significant *P* values are in italic. PC1 and PC2 are scores of the first two axes of a principal components analysis based on the matrix of all plant species cover at each transect
ns not significant tests

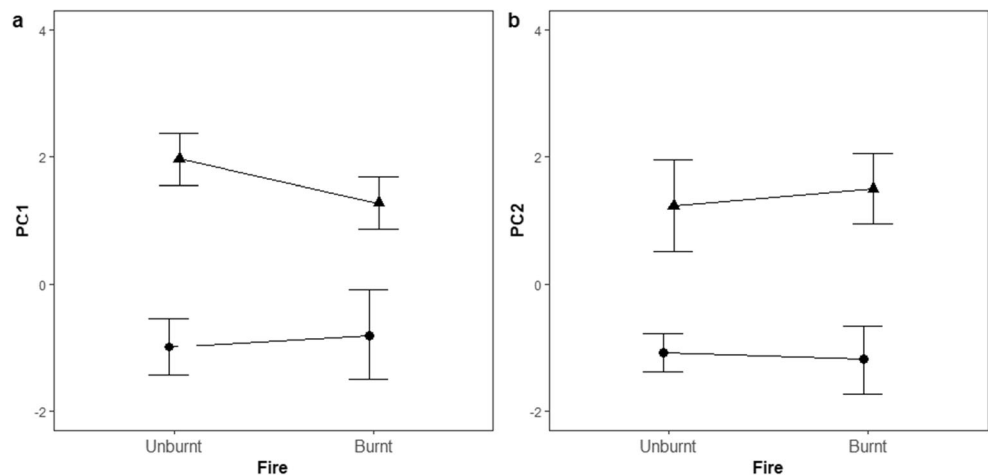
Fig. 2 Generalized linear mixed models and general linear models results of the interaction between forest types and fire in context of **a** canopy, **b** shrub cover, **c** tree height, **d** bare ground cover, **e** litter cover, **f** rock cover, and **g** Simpson index (structure). Circles mean pine plantation; triangles mean cork oak forest. Symbols refer to mean values and whiskers are \pm one standard error



plantations. These divergent results confirm the high post-fire survival ability of *Q. suber* in Western Mediterranean regions such as Spain (Pausas 1997; Gonzalez et al. 2007), Portugal (Moreira et al. 2007; Catry et al. 2009), and Algeria (Bekdouche et al. 2008). It is known that *Q. suber* has thick bark and resprouts vigorously, while fire-created seedbeds are suitable for acorn germination (Pausas 1997). Usually *Q. suber* recovers very quickly because it resprouts directly from

the trunk and branches, this allowing the re-establishment of the canopy in a short period of time (Pausas 1997). Fire vulnerability (and lower resprouting) increases for young or recently debarked *Q. suber* individuals, as well as depending on the location of the forest and timing of the fire (Catry et al. 2012). Despite these differences among populations, there is a high survival rate of this species in fire-prone environments, probably the highest among Western Mediterranean tree

Fig. 3 General linear models results of the interaction between forest types and fire in context of the first two axes PC1 (a) and PC2 (b) of a principal components analysis of the relative abundance of each plant species found in transects. Circles mean pine plantation; triangles mean cork oak forest. Symbols refer to mean values and whiskers are \pm one standard error



species (González et al. 2007). Short intervals between successive fires can lead to substantial changes in vegetation (Lippitt et al. 2012; Tessler et al. 2015), including *Q. suber* forests (Santos and Cheylan 2013). However, repeated fire regimes do not appear to be typical of our study area since most of the study sites have burnt only once over the last 50 years (Chergui et al. 2017).

Our results would support the slow post-fire recovery of pine plantations especially in dry regions during the first years after fire (Trabaud 1982; Domínguez et al. 2002; Perula et al. 2003); this is because most *P. pinaster* individuals die, as do most other coniferous species (Catry et al. 2006; Proença et al. 2010). Pine plantations are highly flammable as they consist of dense stands of pine trees with branches all along the main stem (Pausas et al. 2008) and due to the quality, quantity, and structural arrangement of its fuel (Nimour Nour 1997; Fernandes and Rigolot 2007). Thus, fire progression in the pine plantation canopy is further encouraged by the high tree density (Perula et al. 2003) and the ladder fuel (e.g., lower dead branches, fallen needles) that promote the vertical progression of fire (Fernandes and Rigolot 2007; Fernandes 2009; Ormeño et al. 2009).

The low shrub cover resilience observed in pine plantations can be caused by the slow recovery rate of the dominant tree species and the high intensity of fire, which may destroy regenerative tissues and reduce seed viability (Proença et al. 2010). By contrast, there was similar shrub cover between burnt and unburnt cork oak forest plots, probably due to the post-fire preservation of the extensive root system of shrubs. When the root system is preserved, resprouting can occur immediately after disturbance, taking advantage of the nutrients and water accumulated in the roots (Fernández and Paruelo 1988; Verdú 2000). Contrary to the scrubland, grass cover did not vary after fire in pine plantations and tended to decline in the cork oak forest. The slight effect of fire on grassy species was probably because most post-fire annual species are transitory in these ecosystems (Trabaud and Lepart 1980; Bonnet et al. 2002; Buhk et al. 2006). In addition, the high intensity that characterizes fire events of pine plantations could act as short-term inhibitor of other plant species in these communities.

4.2 Vegetation composition

Our study demonstrated that the composition of plant communities between *Q. suber* and *P. pinaster* woodlands differed. However, the resilience of both forest types to fire was similar under similar fire regime as in the Rif (Chergui et al. 2017). In this scenario, the most evident cause of shift in plant composition is expected to be not fire but the use of conifer plantation. The compositional resilience of plant communities both in native *Q. suber* forests and in *P. pinaster* plantations highlights the evolutionary

strategies of Mediterranean plant species to respond to burning, i.e. resprouting and seeding (Hodgkinson 1998; Pausas and Vallejo 1999; Coca and Pausas 2012; Ne'eman et al. 2012). For example, many shrub species observed in the studied sites were supported mainly by resprouting shoots (James 1984; Canadell and Zedler 1995; Lloret et al. 2003), with fast regeneration processes immediately after fire. This is the case of *Arbutus unedo* (Cabezudo et al. 1995). In addition, post-fire seedlings recruited from dormant soil-stored seed banks (Keeley et al. 2012) are also confirmed in the commonest species recorded in this study such as *Cistus salviifolius* (Trabaud and Oustric 1989; Roy and Sonié 1992). The presence of these species guarantees the resilience in the composition of plant communities from the north-western Africa, at least under a fire regime characterized by few big blazes (> 500 ha) and few areas with recurrent burning (Chergui et al. 2017).

4.3 Conclusions and conservation remarks

In conclusion, *Q. suber* and *P. pinaster* plantations support different plant communities in the African rim of the Western Mediterranean Basin. However, these communities show similar resilience to fire. In contrast, the structure of *Q. suber* stands recover more rapidly after burning than do *Pinus* plantations under a similar fire regime (Calvo et al. 2003). This was caused primarily by the functional strategies of *Q. suber* and *P. pinaster* responding to fire, which may affect other structural layers. Differences in structural resilience between forest types can disturb the resilience of other ecosystem components such as fauna and soil (Seybold et al. 1999; Jacquet and Prodon 2007; Mateos et al. 2011; Chambers et al. 2014). The low post-fire regeneration shown by pine plantations can have direct consequences in the loss of diversity under changes in the current fire regime (Pausas and Fernández-Muñoz 2012; Chergui et al. 2017). The resilience of other components of the ecosystem merits future research to guarantee the conservation of biodiversity in this area characterized by high rate of endemism (Pleguezuelos et al. 2010).

Acknowledgments We wish to thank Professor Mohamed Kadiri, Director of the Applied Botanical Laboratory (Biology Department, Abdelmalek Essaâdi University), for helping on the identification of plant species. Also, we would like to thanks Juli Pausas and Xavier Santos for their revisions of an early version of the manuscript. In addition, we are grateful to Saül Yubero for their kindness and help in the field work.

Funding This study was partially financed by the Instituto de Estudio Ceuties (research grant 2015-1).

Compliance with ethical standards

Conflict of interest The authors declare that they have no conflict of interest.

References

- Ajbilou R (2001) Biodiversidad de los bosques de la península Tingitana (Marruecos). PhD dissertation, University of Sevilla, Sevilla, Spain
- Alanís-Rodríguez E, Jiménez-Pérez J, Valdecantos-Dema A, Pando-Moreno M, Aguirre Calderón O, Treviño-Garza EJ (2011) Caracterización de regeneración leñosa September 2017 postincendio de un ecosistema templado del parque ecológico Chipinque, México. *Rev Chapingo Ser Cienc For Ambient* 17:31–39. <https://doi.org/10.5154/r.rchscfa.2010.05.032>
- Aronson J, Pereira JS, Pausas JG (eds) (2009) Cork oak woodlands on the edge: ecology, adaptive management, and restoration. Island Press, Washington, D.C., p 352
- Barberis A, Dettori S, Filigheddu MR (2003) Management problems in Mediterranean cork oak forests: post-fire recovery. *J Arid Environ* 54:565–569. <https://doi.org/10.1006/jare.2002.1079>
- Bates D, Maechler M, Bolker B, Walker S (2015) Fitting linear mixed-effects models using lme4. *J Stat Softw* 67:1–48
- Bekdouche F, Derridj A, Krouchi F (2008) Evolution après feu de la composition floristique de la subéraie de Mizrana (Tizi-Ouzou, Algérie). *Sci Tech C* 28:19–29
- Belghazi B, Romane F (1994) Productivité du pin maritime (*Pinus pinaster* sol. var. magh.) en peuplements artificiels dans le nord du Maroc. *Rev For Médit t XV* n° 4
- Belghazi B, Ezzahiri M, Romane F (2000) La productivité des peuplements naturels de pin d'Alep dans la forêt de Tamga (Haut Atlas Marocain). *Rev For Médit t XXI* n° 4
- Benabid A (2007) Description de la Composition et de la Structure des Types de Combustibles, « Appui à la mise en œuvre du programme forestier national Elaboration des cartes de risques aux incendies de forêts du Nord du Maroc », 14 pp
- Bond WJ, Woodward FI, Midgley GF (2005) The global distribution of ecosystems in a world without fire. *New Phytol* 165:525–538. <https://doi.org/10.1111/j.1469-8137.2004.01252.x>
- Bonnet V, Dutoit T, Taton T (2002) Spatial gradients of vegetation and soil in the calcareous Provence (France). In: Trabaud L, Prodon R (eds) *Fire and biological processes*. Backhuys Publishers, Leiden, pp 303–314
- Buhk C, Götzenberger L, Wesche K, Sánchez Gómez P, Hensen I (2006) Post-fire regeneration in a Mediterranean pine forest with historically low fire frequency. *Acta Oecol* 30:288–298. <https://doi.org/10.1016/j.actao.2006.05.010>
- Burgman MA, Lamont BB (1992) A stochastic model for the viability of *Banksia cuneata* populations: environmental, demographic, and genetic effects. *J Appl Ecol* 29:719–727. <https://doi.org/10.2307/2404481>
- Burrows ND (2008) Linking fire ecology and fire management in south-west Australian forest landscapes. *For Ecol Manag* 255:2394–2406. <https://doi.org/10.1016/j.foreco.2008.01.009>
- Cabezudo B, Latorre A, Nieto J (1995) After fire regeneration in a *Quercus suber* forest in the south of Spain (Istan, Malaga). *Acta Bot Malacit* 20:143–151
- Calvo L, Santalla S, Marcos E, Valbuena L, Tárrega R, Luis E (2003) Regeneration after wildfire in communities dominated by *Pinus pinaster*, an obligate seeder, and others dominated by *Quercus pyrenaica*, a typical resprouters. *For Ecol Manag* 184:209–223. [https://doi.org/10.1016/S0378-1127\(03\)00207-X](https://doi.org/10.1016/S0378-1127(03)00207-X)
- Canadell J, Zedler PH (1995) Underground structures of woody plants in Mediterranean ecosystems of Australia, California, and Chile. In: Arroyo MTK, Zedler PH, Fox MD (eds) *Ecology and biogeography of Mediterranean ecosystems in Chile, California, and Australia*. Springer, New York, pp 170–210
- Carrión J, Parra I, Navarro C, Munuera M (2000) The past distribution and ecology of the cork oak (*Quercus suber*) in the Iberian Peninsula: a pollen-analytical approach. *Divers Distrib* 6:29–44. <https://doi.org/10.1046/j.1472-4642.2000.00070.x>
- Catry F, Rego FC, Bugalho MN, Lopes T, Silva JS, Moreira F (2006) Effects of fire on tree survival and regeneration in a Mediterranean ecosystem. *For Ecol Manag* 234S:S180–S207. <https://doi.org/10.1016/j.foreco.2006.08.316>
- Catry FX, Moreira F, Duarte I, Acacio V (2009) Factors affecting post-fire crown regeneration in cork oak (*Quercus suber* L.) trees. *Eur J For Res* 128:231–240. <https://doi.org/10.1007/s10342-009-0259-5>
- Catry F, Rego F, Moreira F, Fernandes P, Pausas J (2010) Post-fire tree mortality in mixed forests of Central Portugal. *For Ecol Manag* 260:1184–1192. <https://doi.org/10.1016/j.foreco.2010.07.010>
- Catry FX, Moreira F, Pausas JG, Fernandes PM, Rego F, Cardillo E, Curt T (2012) Cork oak vulnerability to fire: the role of bark harvesting, tree characteristics and abiotic factors. *PLoS One* 7(6):e39810
- Chambers JC, Bradley BA, Brown CA, D'Antonio C, Germino MJ, Hardegree SP, Grace JB, Miller RF, Pyke DA (2014) Resilience to stress and disturbance, and resistance to *Bromus tectorum* L. invasion in the cold desert shrublands of western North America. *Ecosystems* 17:360–375. <https://doi.org/10.1007/s10021-013-9725-5>
- Chergui B, Fahd S, Santos X, Pausas JG (2017) Socioeconomic factors drive fire regime variability in the Mediterranean Basin. *Ecosystems* 21:619–628. <https://doi.org/10.1007/s10021-017-0172-6>
- Clarke PJ, Lawes MJ, Midgley JJ, Lomant BB, Ojeda F, Burrows GE (2013) Resprouting as a key functional trait in woody plants: how buds, protection and resources drive persistence after fire. *New Phytol* 197:19–35. <https://doi.org/10.1111/nph.12001>
- Coca M, Pausas JG (2012) Scale-dependent segregation of seeders and resprouters in cork oak (*Quercus suber*) forests. *Oecologia* 168:503–510. <https://doi.org/10.1007/s00442-011-2102-1>
- Costa A, Madeira M, Santos JL, Oliveira A (2011) Change and dynamics in Mediterranean evergreen oak woodlands landscapes of south-western Iberian Peninsula. *Landsc Urban Plan* 102(3):164–176. <https://doi.org/10.1016/j.landurbplan.2011.04.002>
- Cruz MG, Fernandes PM (2008) Development of fuel models for fire behaviour prediction in maritime pine (*Pinus pinaster* Ait.) stands. *Int J Wildland Fire* 17:194–204. <https://doi.org/10.1071/WF07009>
- DeBano FL, Neary DG, Ffolliott PF (1998) *Fire's effects on ecosystems*. Wiley, New York, p 352
- De Grandpré L, Morissette J, Gauthier S (2000) Long-term post-fire changes in the northeastern boreal forest of Quebec. *J Veg Sci* 11:791–800. <https://doi.org/10.2307/3236549>
- Díaz-Delgado R, Lloret F, Pons X, Terradas J (2002) Satellite evidence of decreasing resilience in Mediterranean plant communities after recurrent wildfires. *Ecology* 83:2293–2303. [https://doi.org/10.1890/00129658\(2002\)083\[2293:SEODRI\]2.0.CO;2](https://doi.org/10.1890/00129658(2002)083[2293:SEODRI]2.0.CO;2)
- Daubenmire R (1959) A canopy-coverage method of vegetation analysis. *North West Sci* 33:43–64
- Dominguez L, Calvo L, Luis E (2002) Impact of wildfire season on regeneration of *Quercus pyrenaica* forest and *Pinus* sp. stand. *J Med Ecol* 3:47–54
- Drever CR, Peterson G, Messier C, Bergeron Y, Flannigan M (2006) Can forest management based on natural disturbances maintain ecological resilience? *Can J For Res* 36:2285–2299. <https://doi.org/10.1139/x06-132>
- Emberger L (1955) Une classification biogéographique des climats. *Recueil des Travaux du Laboratoire de Botanique et Géologie. Faculté des Sciences de Montpellier. Sér Bot* 7:3–43
- Fernandes PM, Rigolot E (2007) The fire ecology and management of maritime pine (*Pinus pinaster* Ait.). *For Ecol Manag* 241:1–13. <https://doi.org/10.1016/j.foreco.2007.01.010>
- Fernandes PM (2009) Combining forest structure data and fuel modeling to classify fire hazard in Portugal. *Ann For Sci* 66:415–424. <https://doi.org/10.1051/forest/2009013>

- Fernández RJ, Paruelo JM (1988) Root systems of two Patagonian shrubs: a quantitative description using a geometrical method. *J Range Manag* 41:220–223. <https://doi.org/10.2307/3899171>
- Folke C (2006) Resilience: the emergence of a perspective for social–ecological systems analysis. *Glob Environ Chang* 16:253–267. <https://doi.org/10.1016/j.gloenvcha.2006.04.002>
- Ghallab A, Taiqui L (2015) Modélisation de la distribution spatiale des paramètres bioclimatiques dans la région Tanger Tétouan (Rif occidental). *Eur Sci J* edition 11, n° 17
- González JR, Trasobares A, Palahí M, Pukkala T (2007) Predicting stand damage and tree survival in burned forests in Catalonia (North-east Spain). *Ann For Sci* 64:733–742. <https://doi.org/10.1051/forest:2007053>
- Hernández-Serrano A, Verdú M, González-Martínez SC, Pausas JG (2013) Fire structures pine serotiny at different scales. *Am J Bot* 100:2349–2356. <https://doi.org/10.3732/ajb.1300182>
- Hodgkinson KC (1998) Sprouting success of shrubs after fire: height-dependent relationships for different strategies. *Oecologia* 115:64–72. <https://doi.org/10.1007/s004420050492>
- Jacquet K, Prodon R (2007) Résilience comparée des peuplements de chêne vert et de chêne-liège après incendie. *Rev For Fr* 59:31–44.
- James S (1984) Lignotubers and burls—their structure, function and ecological significance in Mediterranean ecosystems. *Bot Rev* 50:225–266. <https://doi.org/10.1007/BF02862633>
- Keeley JE (1986) Resilience of Mediterranean shrub communities to fire. In: Dell B, Hopkins AJM, Lamont BB (eds) *Resilience in Mediterranean-type ecosystems*. Dr W Junk, Dordrecht, the Netherlands, pp 95–112
- Keeley JE (2012) Ecology and evolution of pine life histories. *Ann For Sci* 69:445–453. <https://doi.org/10.1007/s13595-012-0201-8>
- Keeley JE, Bond WJ, Bradstock RA, Pausas JG, Rundel PW (2012) *Fire in Mediterranean ecosystems: ecology, evolution and management*. Cambridge University Press, Cambridge
- Lemmon P (1957) A new instrument for measuring forest overstory density. *J For* 55:667–668
- Lipoma ML, Gurvich DE, Urcelay C, Díaz S (2016) Plant community resilience in the face of fire: experimental evidence from a semi-arid shrubland. *Austral Ecology* 41:501–511. <https://doi.org/10.1111/aec.12336>
- Lippitt CL, Stow DA, O’Leary JF, Franklin J (2012) Influence of short-interval fire occurrence on post-fire recovery of fire-prone shrublands in California, USA. *Int J Wildland Fire* 22:184–193. <https://doi.org/10.1071/WF10099>
- Lloret F (1998) Fire, canopy cover and seedling dynamics in Mediterranean shrubland of northeastern Spain. *J Veg Sci* 9:417–430. <https://doi.org/10.2307/3237106>
- Lloret F, Pausas JG, Vilà M (2003) Responses of Mediterranean plant species to different fire frequencies in Garraf Natural Park (Catalonia, Spain): field observations and modelling predictions. *Plant Ecol* 167:223–235. <https://doi.org/10.1023/A:1023911031155>
- Mateos E, Santos X, Pujade-Villar J (2011) Taxonomic and functional responses to fire and post-fire management of a Mediterranean Hymenoptera community. *Environ Manag* 48:1000–1012. <https://doi.org/10.1007/s00267-011-9750-0>
- Mharzi Alaoui H, Assali F, Rouchdi M, Lahssini S, Tahiri D (2015) Analyse de l’interaction entre l’éclosion des feux de forêts et les types de bioclimat au Nord du Maroc—cas de la région du Rif occidental. *Rev Mar Sci Agron Vét* 3:46–53
- Mharzi Alaoui H, Assali F, Rouchdi M, Tahiri D, Lahssini S, Aafi A, Moukrim S (2017) Cartographie de l’intensité du feu et de la sensibilité aux feux de cimes en forêts Méditerranéennes. Cas de la province de Chefchaouen au nord du Maroc. *Rev Ecol Terre Vie* 72:387–409
- Molina JR, Rodriguez Silva F, Herrera MA (2011) Potential crown fire behaviour in *Pinus pinea* stands following different fuel treatments. *Forest Systems* 20:266–277
- Molinas ML, Verdager D (1993) Lignotuber ontogeny in the cork-oak (*Quercus suber*; Fagaceae). *Am J Bot* 80:182–191. <https://doi.org/10.2307/2445038>
- Moreira F, Duarte L, Catty F, Acacio V (2007) Cork extraction as a key factor determining post-fire cork oak survival in a mountain region of southern Portugal. *For Ecol Manag* 253:30–37. <https://doi.org/10.1016/j.foreco.2007.07.001>
- Moreira F, Rego FC, Ferreira PG (2001) Temporal (1958–1995) pattern of change in a cultural landscape of northwestern Portugal: implications for fire occurrence. *Landsc Ecol* 16:557–567. <https://doi.org/10.1023/A:1013130528470>
- Moreira F, Russo D (2007) Modelling the impact of agricultural abandonment and wildfires on vertebrate diversity in Mediterranean Europe. *Landsc Ecol* 22:1461–1476. <https://doi.org/10.1007/s10980-007-9125-3>
- Moritz MA, Parisien MA, Battlori E, Krawchuk MA, Van Dorn J, Ganz DJ, Hayhoe K (2012) Climate change and disruptions to global fire activity. *Ecosphere* 3(6):49. <https://doi.org/10.1890/ES11-00345.1>
- Ne’eman G, Lev-Yadun S, Arianoutsou M (2012) Fire-related traits in Mediterranean Basin plants. *Isr J Ecol Evol* 58:177–194. <https://doi.org/10.1560/IJEE.58.2-3.177>
- Nimour Nour E (1997) *Inflammabilité de la végétation méditerranéenne*. Thesis report. Aix-Marseille University, Marseille, France
- Ormeño E, Céspedes B, Sanchez IA, Velasco-García A, Moreno JM, Fernandez C, Baldy V (2009) The relationship between terpenes and flammability of leaf litter. *For Ecol Manag* 257:471–482. <https://doi.org/10.1016/j.foreco.2008.09.019>
- Pastor-López A, Taiqui L, Bouziane H, Riadi H, Martín JM (1997) Structure of *Quercus suber* forests in Chefchaouen basin (NE. Morocco). Implications on management at a landscape scale. *Mediterránea* 16:65–76
- Pausas JG (1997) Resprouting of *Quercus suber* in NE Spain after fire. *J Veg Sci* 8:703–706. <https://doi.org/10.2307/3237375>
- Pausas JG (1999) Response of plant functional types to changes in the fire regime in Mediterranean ecosystems. A simulation approach. *J Veg Sci* 10:717–722. <https://doi.org/10.2307/3237086>
- Pausas JG (2001) Resprouting vs seeding—a Mediterranean perspective. *Oikos* 94:193–194. <https://doi.org/10.1034/j.1600-0706.2001.t01-1-10979.x>
- Pausas JG, Alessio G, Moreira B, Corcobado G (2012) Fires enhance flammability in *Ulex parviflorus*. *New Phytol* 193:18–23. <https://doi.org/10.1111/j.1469-8137.2011.03945.x>
- Pausas JG, Bladé C, Valdecantos A, Seva JP, Fuentes D, Allozal JA, Vilagrosa A, Bautista S, Cortina J, Vallejo R (2004) Pines and oaks in the restoration of Mediterranean landscapes of Spain: new perspectives for an old practice—a review. *Plant Ecol* 171:209–220. <https://doi.org/10.1023/B:VEGE.0000029381.63336.20>
- Pausas JG, Carbó E, Caturla RN, Gil JM, Vallejo R (1999) Post-fire regeneration patterns in the eastern Iberian Peninsula. *Acta Oecol* 20:499–508. [https://doi.org/10.1016/S1146-609X\(00\)86617-5](https://doi.org/10.1016/S1146-609X(00)86617-5)
- Pausas JG, Fernández-Muñoz S (2012) Fire regime changes in the Western Mediterranean Basin: from fuel-limited to drought-driven fire regime. *Clim Chang* 110:215–226. <https://doi.org/10.1007/s10584-011-0060-6>
- Pausas JG, Llovet J, Rodrigo A, Vallejo R (2008) Are wildfires a disaster in the Mediterranean basin? A review. *Int J Wildland Fire* 17:713–723. <https://doi.org/10.1071/WF07151>
- Pausas JG, Keeley JE (2014) Evolutionary ecology of resprouting and seeding in fire-prone ecosystems. *New Phytol* 204:55–65. <https://doi.org/10.1111/nph.12921>
- Pausas JG, Vallejo VR (1999) The role of fire in European Mediterranean ecosystems. In: Chuvieco E (ed) *Remote sensing of large wildfires in the European Mediterranean basin*. Springer, Berlin, pp 3–16

- Perula VG, Cerrillo RMN, Rebollo PF, Murillo GV (2003) Postfire regeneration of *Pinus pinea* L. and *Pinus pinaster* Aiton in Andalusia (Spain). *J Environ Manag* 31:86–99. <https://doi.org/10.1007/s00267-002-2786-4>
- Pleguezuelos JM, Brito JC, Fahd S, Feriche M, Mateo JA, Moreno-Rueda G, Reques R, Santos X (2010) Setting conservation priorities for the Moroccan herpetofauna: the utility of regional red lists. *Oryx* 44: 501–508. <https://doi.org/10.1017/S0030605310000992>
- Proença VM, Pereira HM, Vicente L (2010) Resistance to wildfire and early regeneration in natural broadleaved forest and pine plantation. *Acta Oecol* 36:626–633. <https://doi.org/10.1016/j.actao.2010.09.008>
- Quezel P, Barbero M, Benabid A, Rivas-Martinez S (1992) Contribution a l'étude des groupements forestiers et pré-forestiers du Maroc oriental. *Stud Bot* 10:57–90
- R Core Team (2015) R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria <http://www.R-project.org>
- Retana J, Espelta JM, Habrouk A, Ordonez JL, de Sola-Morales F (2002) Regeneration patterns of three Mediterranean pines and forest changes after a large wildfire in northeastern Spain. *Ecoscience* 9: 89–97. <https://doi.org/10.1080/11956860.2002.11682694>
- Rodrigo A, Retana J, Picó X (2004) Direct regeneration is not the only response of Mediterranean forests to large fires. *Ecology* 85:716–729. <https://doi.org/10.1890/02-0492>
- Rodríguez-García E, Bravao F, Spies TA (2011) Effects of overstorey canopy, plant plant interactions and soil properties on Mediterranean maritime pine seedling dynamics. *For Ecol Manag* 262:244–251. <https://doi.org/10.1016/j.foreco.2011.03.029>
- Roy J, Sonie L (1992) Germination and population dynamics of *Cistus* species in relation to fire. *J Appl Ecol* 29:647–655. <https://doi.org/10.2307/2404472>
- Santos X, Cheylan M (2013) Taxonomic and functional response of a Mediterranean reptile assemblage to a repeated fire regime. *Biol Conserv* 168:90–98. <https://doi.org/10.1016/j.biocon.2013.09.008>
- Shakesby RA, Boakes D, Coelho COA, Gonçalves AJB, Walsh RPD (1996) Limiting the soil degradation impacts of wildfire in pine and *Eucalyptus* forest in Portugal: a comparison of alternative post-fire management practices. *Appl Geogr* 16:337–355. [https://doi.org/10.1016/0143-6228\(96\)00022-7](https://doi.org/10.1016/0143-6228(96)00022-7)
- Seybold CA, Herrick JE, Brejda JJ (1999) Soil resilience: a fundamental component of soil quality. *Soil Sci* 164:224–234. <https://doi.org/10.1097/00010694-199904000-00002>
- Tapias R, Climent J, Pardos JA, Gil L (2004) Life histories of Mediterranean pines. *Plant Ecol* 171:53–68. <https://doi.org/10.1023/B:VEGE.0000029383.72609.f0>
- Tessler N, Sapir Y, Wittenberg L, Greenbaum N (2015) Recovery of Mediterranean vegetation after recurrent Forest fires: insight from the 2010 forest fire on Mount Carmel, Israel. *Land Degrad Develop* 27:1424–1431. <https://doi.org/10.1002/ldr.2419>
- Thanos CA, Doussi MÁ (2000) Post-fire regeneration of *Pinus brutia* forests. In: Ne' eman G, Trabaud L (eds) Ecology, biogeography and management of *Pinus halepensis* and *P. brutia* forest ecosystems in the Mediterranean Basin. Backhuys Publishers, Leiden, pp 291–301
- Torres I, Pérez B, Quesada J, Viedma O, Moreno JM (2016). Forest shifts induced by fire and management legacies in a *Pinus pinaster* woodland. *For Ecol Manag* 361:309–317. <https://doi.org/10.1016/j.foreco.2015.11.027>
- Trabaud L (1982) Effects of past and present fire on the vegetation of the FrenchMediterranean region. In: Conrad CE, Oechel WC (eds) Proceedings of the symposium on the dynamics and management of Mediterranean-type ecosystems. June, pp 22–26
- Trabaud L, Lepart J (1980) Diversity and stability in garrigue ecosystems after fire. *Vegetatio* 43:49–57
- Trabaud L, Oustric J (1989) Comparison between post-fire regeneration strategies of two *Cistus* species. *Rev Ecol (Terre et Vie)* 44:3–14
- Úbeda X, Outeiro LR, Sala M (2006) Vegetation regrowth after a differential intensity forest fire in a Mediterranean environment, Northeast Spain. *Land Degrad Develop* 17:429–440. <https://doi.org/10.1002/ldr.748>
- Valdecantos A, Baeza MJ, Vallejo VR (2009) Vegetation management for promoting ecosystem resilience in fire-prone Mediterranean shrublands. *Restor Ecol* 17(3):414–421. <https://doi.org/10.1111/j.1526-100X.2008.00401.x>
- Vallejo R, Aronson J, Pausas JG, Cortina J (2006) Mediterranean woodlands. In: van Andel J, Aronson J (eds) Restoration ecology: the new frontier. Blackwell Science, Oxford, pp 193–207
- Vega JA, Fernández C, Pérez-Gorostiaga P, Fonturbel T (2010) Response of maritime pine (*Pinus pinaster* Ait.) recruitment to fire severity and post-fire management in a coastal burned area in Galicia (NW Spain). *Plant Ecol* 206:297–308. <https://doi.org/10.1007/s11258-009-9643-y>
- Verdú M (2000) Ecological and evolutionary differences between Mediterranean seeders and resprouters. *J Veg Sci* 11:265–268. <https://doi.org/10.2307/3236806>
- Walker XJ, Mack MC, Johnstone JF (2016) Predicting ecosystem resilience to fire from tree ring analysis in black spruce forests. *Ecosystems* 20:1137–1150. <https://doi.org/10.1007/s10021-016-0097-5>
- Wickham H (2009) ggplot2: elegant graphics for data analysis. Springer-Verlag, New York