



# Current and future estimates for the fire frequency and the fire rotation period in the main woodland types of peninsular Spain: a case-study approach

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## Abstract

**Aim of study:** Fire regimes are frequently dynamic and change as a function of the interactions between the three main fire drivers: fuels, ignitions and climatic conditions. We characterized the recent period (1974-2005) and performed estimates for the future fire regime.

**Area of study:** We have considered five pine and another four woodland types by means of the analyses of 100 reference areas in peninsular Spain.

**Material and Methods:** The estimates of the expected alterations in fire frequency and the fire rotation period were based on models previously developed for the climatic scenarios SRES A2 and B2.

**Main results:** The results point to the large variability in fire frequency and rotation periods between the woodland types as defined, and also among the reference areas delimited for each of them. Fire frequencies will increase for all woodland types while very relevant shortenings of the fire rotation periods are expected. For the 32 yr period analysed, rotation periods longer than 500 yr were obtained in 54% of the reference areas while this percentage would decrease to 31% in the B2 and to 29% in the A2 climatic scenario. In the most affected woodland type, *P. pinaster*, from a median rotation period of 83 yr it would decrease to 26 yr in the B2 and to 20 yr in the A2 climatic scenario.

**Research highlights:** We conclude that the predicted increases in fire activity will have adverse effects on some of the main Spanish woodland types due to the expected future disruptions in the fire regime.

**Keywords:** Forest fires; fire regime; fire frequency; fire rotation period; climatic change.

**Abbreviations used:** SRES: Special Report on Emissions Scenarios; IPCC: Intergovernmental Panel on Climate Change; RA: Reference Areas.

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## Introduction

Wildland fires are a global ecological disturbance process with complex spatial and temporal distributions. In recent years we are acquiring a greater knowledge about its global distribution (Bond & Keeley, 2005; Krawchuk *et al.*, 2009; Archibald *et al.*, 2013) thanks, to a greater extent, to global remote sensing products (e.g. Riaño *et al.*, 2007) and we are now able to propose a hypothesis about global activity patterns (Krawchuk & Moritz, 2011; Pausas & Ribeiro, 2013).

Wildland fire activity can be characterized by its fire regime, that is the type of fire history registered in a defined space and time. Fire regimes result from the interactions between climate, fuels and ignitions as their three main drivers (e.g. Davis & Burrows, 1994; Moreno *et al.*, 1998). Fire regime is determined by a large number of factors such as the presence of ignition sources, the topography, prevention and extinction efficiency, existence of climatic conditions which favour the spread of fire and the amount, type and arrangement of fuels at several spatial scales (Rothermel, 1983). The fire regime represents a particular combination of fire

characteristics such as frequency, intensity, size, season, type and extent (e.g. Trabaud, 1981; Bond & van Wilgen, 1996; Archibald *et al.*, 2013).

Fire regimes frequently change in time and space, not only because of the fortuitousness of fire events (Moritz *et al.*, 2012; San-Miguel-Ayanz *et al.*, 2013) at short spatial and temporal scales, but also as a function of the interactions between the main fire drivers: fuels, ignitions and climatic conditions. This variability in drivers is exacerbated in the context of global change in which we are now (IPCC, 2007; 2014). This has been the case in southern European countries in which all the three fire drivers have been changing quickly since the middle of the past century leading to strong changes in fire regime (Le Houerou, 1987; Moreno *et al.*, 1998; Pausas & Fernandez-Muñoz, 2012; San-Miguel-Ayanz *et al.*, 2013; Moreno & Chuvieco, 2013).

The woodland area burned accounted for 37% (see results) of the total forest area affected by fire in peninsular Spain in the period analysed (1974-2005) while the rest were scrublands, pastures and cultivated areas. Nevertheless, the ecological relevance of ecosystems dominated by trees, its economic importance as ecosystem services suppliers and the possibility of identification of the affected woody species in the available fire reports support the analyses of the fire regime in different woodland types in this paper. Additionally, fire regimes for woodland types with a large distribution area could be very variable depending on the climatic, landscape and cultural differences within that area (e.g. Trabaud, 1994; Moreno *et al.*, 1998; Díaz-Delgado *et al.*, 2004; Moreira *et al.*, 2011; Vila-Cabrera *et al.*, 2012). In general, the spatial distribution of different types of forest over large areas is mainly conditioned by climatic gradients which control primary productivity and the structure of vegetation, but the spatial distribution of woodland types and structures in anthropogenic, or man-made, landscapes as those found in the Mediterranean basin is largely dependent on human activities and can determine also the behaviour of fire (e.g. Ruiz-Benito *et al.*, 2012; Alvarez *et al.*, 2012).

In addition to the availability of fuel to carry the fire and the ignition sources requirements, the climatic/weather conditions determine to a large extent fire activity. Several studies have indicated that large percentages of variance in fire activity can be explained as a function of simple climatic variables (e.g. Piñol *et al.*, 1998; Pausas, 2004; Flannigan *et al.*, 2009; Koutsias *et al.*, 2013). According to the IPCC (2007; 2014) reports, the risk of fire, length of the fire season, fire frequency and severity are very likely to increase in Mediterranean type ecosystems. Climatic conditions which favour the spread of fire will become more frequent in the Mediter-

ranean basin (Moriondo *et al.*, 2006; Lung *et al.*, 2013) while several studies concerning the Iberian Peninsula, point to an increment in fire risk under future climate scenarios (e.g. Moreno, 2005; Carvalho *et al.*, 2010, Vázquez *et al.*, 2012). This possible increase in fire activity could have important effects on forest landscapes (e.g. Schumacher & Bugmann, 2006). However, in spite of the key role of fire in forest distribution and structure there is still a lack of knowledge of current fire activity in specific woodland types and also a lack of estimates for future climatic scenarios.

In order to improve this knowledge, the strategy followed in this work was based on the delimitation of several Reference Areas (RA) or case-studies for each of the nine woodland types considered. In this way we can evaluate the current fire regime registered, and perform future estimates, for each of the species considered. With this approach we hope to provide a picture of the broad range of variability in current and also in future estimates. To quantify the expected alterations in future fire regime parameters we have used statistical models previously published (Vázquez *et al.*, 2012). The models used were based in the relationships between fire regime parameters and monthly meteorological variables for several ecozones in which peninsular Spain was divided up for the climatic scenarios (Nakicenovic *et al.*, 2000) SRES A2 and B2 (2071-2100).

We have three main objectives in this work. The first one is to provide a characterization of the fire frequency and the fire rotation period registered in the main woodland types defined and its spatial variability by means of the reference areas (RA) considered. The second is to explore the differences in these parameters between the main pine species and the third to perform estimates of future fire regime parameters such as the fire frequency and the fire rotation period in the RA defined for the nine woodland types. Our main hypothesis is that fire regime parameters will be very different between the woodland types considered and also among the reference areas considered for each of the woodland types. Additionally we expect a higher fire activity in some of the woodland types more used in afforestation and reforestation programs and large differences between the different pine woodlands that constitute five of the nine woodland types considered.

## Material and methods

### Forest fire data and woodland types

The forest fire data were provided by the Spanish Forest Administration, covering the 32 yr period 1974-2005. Each fire record was referenced to the 10x10 km

grid-unit based on the UTM projection in which the fire started. Reliable data were not available for the whole period for two of the 47 provinces (Álava and Navarra) in peninsular Spain, so they were excluded in the analyses. Insular provinces were also excluded. Raw (1974-1990) and elaborated (1991-2005) data were screened and checked against the published official fire statistics. Only fires with a reported size  $\geq 0.1$  ha of burned area are considered. When a fire burns woodlands or dense forest areas, each of the combinations of the tree species affected, the stand development or the type of property (e.g. public or private) of the area affected was recorded individually along with area burned. Based on this information we have defined the “number of cases” as the number of times each of the nine woodland types considered has been affected by fire. We have used the number of cases for each woodland type as a surrogate for computing the fire frequency in each woodland type.

The woodland (or dense forest) types considered are made up by the five main pine species (*Pinus sylvestris* L., *P. pinea* L., *P. halepensis* Mill., *P. nigra* Arn. and *P. pinaster* Ait.), the genera *Eucalyptus* spp. and *Quercus* spp. and two miscellaneous types grouped into “other conifers” and “other broadleaves”. In this way we have grouped the trees affected by fire into nine types with an emphasis on the main *Pinus* spp. growing in peninsular Spain. These will be named as woodland types, in spite of the fact that they could be dense forests, as opposed to scrublands (Table 1).

## Reference areas (RA) and current fire regime

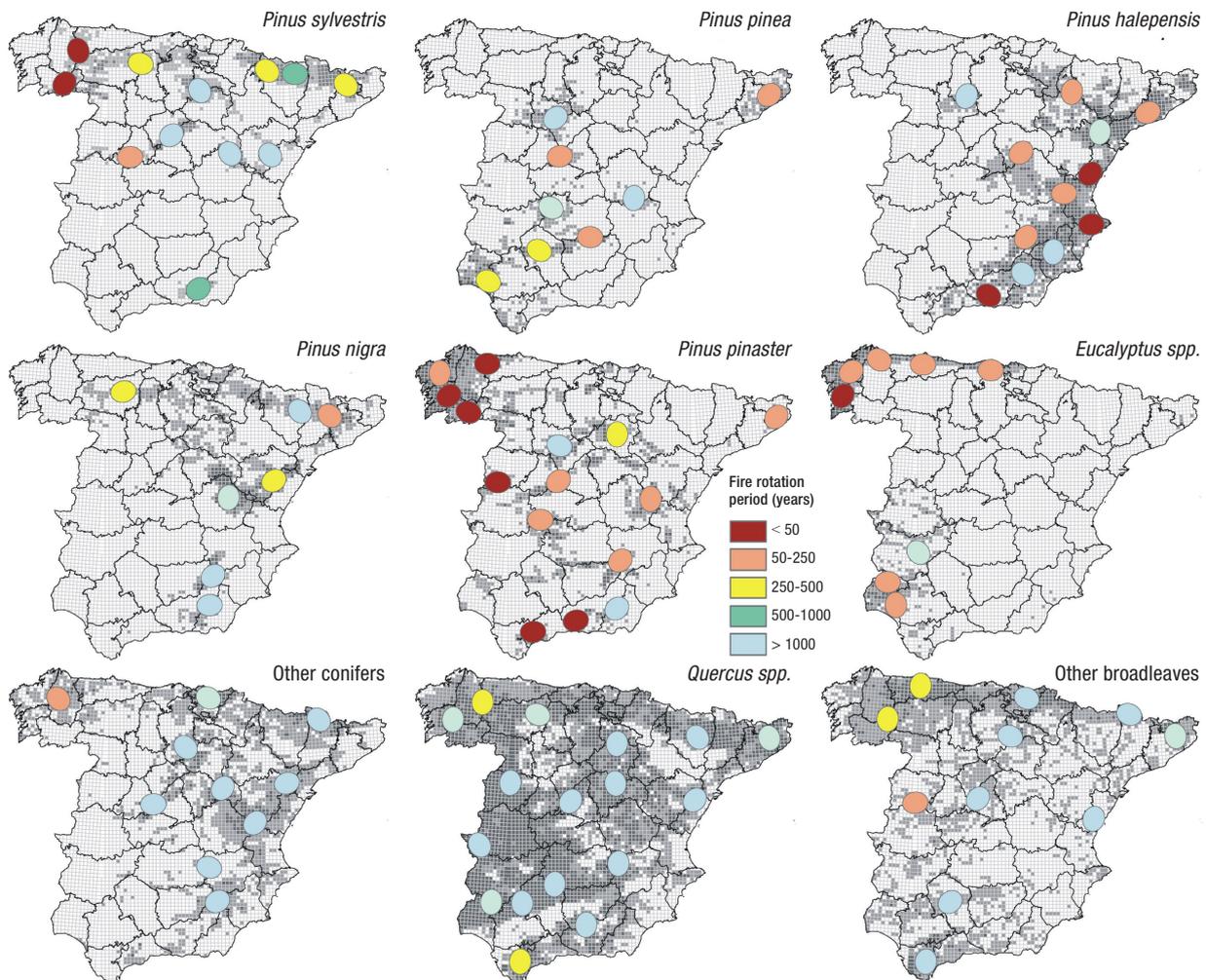
We have defined a total of 100 reference areas (RA) or case-studies to get an idea of the variability in the fire activity for each woodland type. The selected reference areas and the distribution of the woodland types considered are displayed in Fig. 1. The area covered by each woodland type, represented in grey tones in Fig. 1, has been derived from the Forest Map of Spain (MFE) at a 1:50.000 scale. In order to compare this information with the fire data, we have calculated for each of the 5007 10x10 grid-units used as reference for fire location in peninsular Spain, the area covered by each of the woodland types. The total size of each RA is very similar and close to 300,000 ha. The shape is elliptical and they are rotated to encompass in each of them a large area of each woodland type to which the RA is dedicated, that is each RA is built for a specific woodland type. It is important to note that in the process of selection and location of the RA we considered only the species distribution, disregarding forest fire activity or area burned. The RA are then built as case-studies of the main distribution areas of each woodland type and are located in regions with a good representation of the species of interest. More details of the RA are shown in Table 2 and 3 and Fig. 2 and shown later.

We used two fire regime parameters to assess the variability in the current fire regime in the nine types of woodlands and to estimate its future variability. The first one is fire frequency defined as the number of fire cases in 10,000 ha of wooded forest surface

**Table 1.** Wooded forest area (based on the MFE50 Forest Map of Spain) and number of grid-units with  $\geq 100$  ha of the woodland type defined. The proportion (%) of grid-units affected by fire is calculated from grid-units with  $\geq 100$  ha of forest area and  $\geq 100$  ha of area burned for each of the woodland types. Number of cases and area burned by the fires reported for the nine woodland types considered in peninsular Spain for the period 1974-2005. The proportions (%) for the number of cases and the woodland area burned make 100%. Only fires with a reported size  $\geq 0.1$  are considered. The two provinces (Álava and Navarra) with incomplete data are excluded giving a total of 4864 grid-units

Woodland type	Wooded Forest area (MFE50)		Grid-units affected	Number of cases (1974-2005)		Woodland area burned (1974-2005)	
	(ha)	# grid-units ( $\geq 100$ ha)	% ( $\geq 100$ ha burned)	cases	(%)	(ha)	(%)
<i>Pinus sylvestris</i>	999,154	798	37.0	8,222	4.4	177,476	8.1
<i>Pinus pinea</i>	439,148	582	18.9	4,669	2.5	65,114	3.0
<i>Pinus halepensis</i>	1,831,316	1,333	29.7	17,208	9.2	524,139	23.8
<i>Pinus nigra</i>	782,122	765	14.8	3,534	1.9	91,448	4.1
<i>Pinus pinaster</i>	1,287,380	1,181	52.2	63,532	33.9	695,931	31.6
<i>Eucalyptus</i> spp.	621,447	559	53.0	36,373	19.4	224,390	10.2
Other conifers	1,213,261	1,621	15.4	14,181	7.6	105,933	4.8
<i>Quercus</i> spp.	7,557,678	3,696	14.0	26,253	14.0	269,123	12.2
Other broadleaves	1,196,872	2,023	6.7	13,391	7.1	50,494	2.3
Total				187,363	100.0	2,204,050	100.0

## Current fire regime (1974-2005)



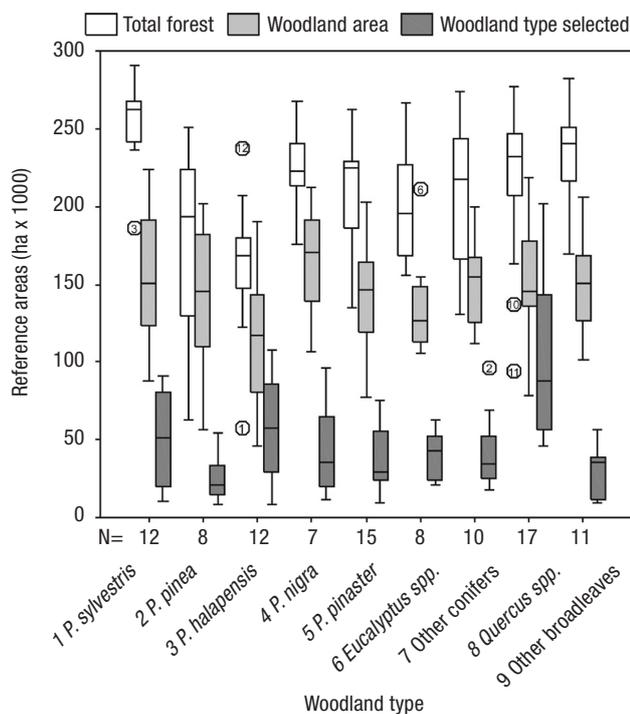
**Figure 1.** Maps displaying the location of the RA defined for the nine woodland types and the fire rotation period calculated for the current period (1974-2005). All the reference areas are of similar size (close to 300,000 ha) and shape but they can change in alignment. Grey tones indicate the proportion of area covered by the woodland type of interest in each 10 x 10 km grid-unit: light grey between 100 and 1000 ha and dark grey more than 1000 ha.

**Table 2.** Characterization of the Reference Areas (RA) selected. Number of RA, wooded forest area (based on the MFE50 Forest Map of Spain), number of cases and area burned for the RA selected for each woodland type. The proportions (%) for the wooded forest area, the number of cases and the woodland area burned are calculated between the values reported in the table and the totals for peninsular Spain displayed in Table 1. Only fires with a reported size  $\geq 0.1$  are considered

Woodland type	Number of RA	Wooded Forest area (MFE50)		Number of cases (1974-2005)		Woodland area burned (1974-2005)	
		(ha)	%	cases	%	(ha)	%
<i>Pinus sylvestris</i>	12	588,315	58.9	2,828	34.4	69,213	39.0
<i>Pinus pinea</i>	8	202,009	46.0	1,777	38.1	21,822	33.5
<i>Pinus halepensis</i>	12	698,044	38.1	6,873	39.9	242,998	46.4
<i>Pinus nigra</i>	7	312,741	40.0	881	24.9	39,937	43.7
<i>Pinus pinaster</i>	15	551,554	42.8	21,121	33.2	321,034	46.1
<i>Eucalyptus spp.</i>	8	321,214	51.7	18,822	51.7	119,226	53.1
Other conifers	10	418,489	34.5	2,644	18.6	12,511	11.8
<i>Quercus spp.</i>	17	1,749,047	23.1	3,948	15.0	44,114	16.4
Other broadleaves	11	316,517	26.4	2,914	21.8	13,535	26.8
All RA types	100	5,157,931	32.4	61,808	33.0	884,390	40.1

**Table 3.** Mean and standard error for the elevations and a climatic characterization of the RA selected for each woodland type. The observed (Ninyerola *et al.*, 2005) precipitation and temperature and the modelled temperatures for the control period (1961-1990) and the climatic scenarios B2 and A2 (2071-2100) derived from the PROMES Regional Climate Model (AEMET 2009) are provided

Woodland type RA	Elevation		Observed precipitation and temperatures		Estimated temperatures (°C, AEMET)		
	Mean	St. error	mm	°C	Control	B2	A2
<i>Pinus sylvestris</i>	1,118.4	50.4	844.6	10.4	9.4	12.5	13.5
<i>Pinus pinea</i>	602.7	95.7	597.0	14.4	14.0	17.3	18.4
<i>Pinus halepensis</i>	697.6	71.2	508.5	13.8	13.0	16.2	17.2
<i>Pinus nigra</i>	1,044.2	65.5	619.3	11.5	10.9	14.1	15.1
<i>Pinus pinaster</i>	745.4	73.0	833.7	12.9	12.2	15.2	16.3
<i>Eucalyptus spp.</i>	374.4	43.0	1,111.6	13.8	13.8	16.5	17.6
Other conifers	935.9	124.3	749.4	11.6	10.4	13.5	14.6
<i>Quercus spp.</i>	736.3	64.8	744.2	13.1	12.6	15.7	16.8
Other broadleaves	753.9	134.8	899.4	12.3	11.5	14.5	15.6



**Figure 2.** Box-plots displaying the range of values in the total forest area (woodland and shrublands), the total woodland area (of the nine woodland types) and the area of the woodland type selected for which the RA is dedicated. The number of RA (reference areas or case-studies) is also displayed. The data are derived from the Forest Map of Spain.

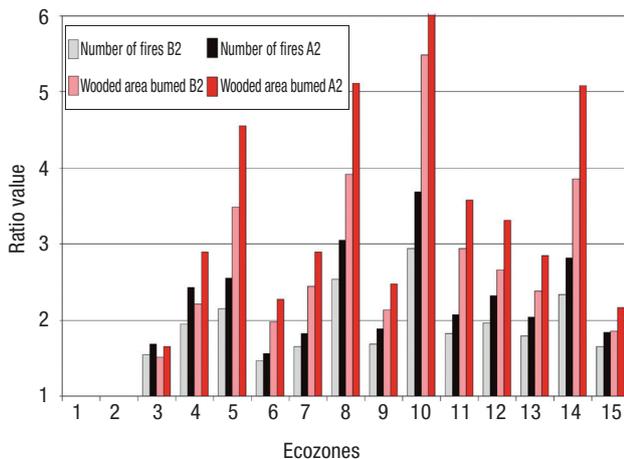
and year. As explained before, we used the number of cases in each woodland type as a surrogate for fire incidence. The number of years with information was assumed to be 32 in all the grid-units in spite of some missing information. The second fire regime param-

eter is the fire rotation period defined as the number of years needed to burn an area equivalent to the one used as reference (Heinselman, 1973). In the same way as for fire frequency the surfaces involved were only those of the woodland type for which we are calculating the fire rotation period. The fire rotation period calculated in this not spatially explicit way, did not take into account the recurrence of fires in the same places.

### Future fire regime parameters estimates

We estimated the future fire incidence in each RA by applying to each one a “ratio value” according to the ecozone in which it is located (Vázquez *et al.*, 2012). Ecozones are based on the concept of potential natural vegetation. The map of Vegetation Series for Spain (Rivas-Martínez 1987) was used to classify each 10x10 km grid-unit according to the dominant vegetation type that was then assigned to the ecozones considered. Ecozones 1 to 6 are dominated by mountain conifers (*Pinus sylvestris*, *P. uncinata*, *Abies alba*, *Juniperus thurifera*) and deciduous trees (*Fagus sylvatica*, *Quercus robur*, *Q. pyrenaica*, *Q. faginea*). Ecozones 7 to 13 are dominated by sclerophyllous species, among which *Q. rotundifolia* is prevalent. *Q. suber* dominates type 14 areas and type 15 areas are dominated by *Q. coccifera*. Although conifers (mainly pine species) represent a high proportion of the current forested area of Spain, on this map they are mostly considered as subdominant in other vegetation units, forming secondary forests or increasing in extension as a result of afforestation.

Ratio values are thought to represent the proportion between current (1974-2005) and future (2071-2100) fire incidence (Vázquez *et al.*, 2012), and were based on linear regressions between monthly temperatures and fire activity variables. The ratio applied to each RA for obtaining future estimates was based on the ratio value of the ecozone in which the RA is located (Fig. 3). From the 100 RA selected we got estimates for 97 since no valid models were obtained for ecozones 1 and 2. The models and limitations were more deeply explained in Vázquez *et al.* (2012). We assumed that the ratios obtained for the different ecozones for the number of fires and for the wooded area burned can be applied equally to the number of cases, that is the number of times each of the woodland types has been affected by fire and to the area burned by each of them. Future scenarios were based on the anthropogenic emission estimates of the IPCC SRES (Special Report on Emissions Scenarios) A2 and B2 (Nakicenovic *et al.*, 2000). In both SRES A2 (high emission) and B2 (low emission) scenarios Europe would undergo a warming in all seasons.



**Figure 3.** Ratios values estimated between current fire incidence (number of fires and wooded area burned) and the estimates for the SRES climatic scenarios B2 and A2. No valid models were obtained for ecozones 1 and 2. Data based on Vázquez *et al.* 2012.

## Results

### Forest fires in the period 1974-2005 in peninsular Spain

In the period 1974-2005 wildland fires affected to close to  $2.2 \times 10^6$  ha of woodland forest area, which represented just 37% of the total area burned in this period that can be quantified in more than  $5.9 \times 10^6$  ha

in 343,723 fires with a reported size  $\geq 0.1$  ha. The five species of *Pinus* accounted for 34% of the wooded forest area in peninsular Spain but were implicated in 52% of the number of cases and 71% of the wooded forest area burned. The remaining tree species integrated in the other four wildland types accounted for more a greater wooded forest area (66%) and were implicated in less cases (48%) and in much less area burned (29%) (Table 1). A large proportion of the number of cases is due to *P. pinaster* (33.9%) followed by *Eucalyptus spp.* (19.4%) and *Quercus spp.* (14%). In relation to the woodland area burned the largest proportion is due also to *P. pinaster* (31.6%) but *P. halepensis* (23%) is in the second position. The *Pinus* species, except *P. pinaster*, have greater proportions in the area burned than in the number of cases. The opposite is observed in the remaining woodland types indicating smaller fires. The proportion of grid-units affected by fire, taking into account only the grid-units with more  $\geq 100$  ha of wooded forest area as displayed in Fig. 1 and also  $\geq 100$  ha of wooded area burned, is in general greater in *Pinus*. The largest value is obtained in *Eucalyptus spp.* (53%) followed by *P. pinaster* (52.2%), *P. sylvestris* (37%) and *P. halepensis* (29.7%) (Table 1).

### Reference areas (RA)

The number of RA selected ranged from 7 for *P. nigra* to 17 for *Quercus spp.* These 100 reference areas (RA) encompassed 32.4% of the area covered by wooded forest in peninsular Spain ranging from 202,009 ha in *P. pinea* to 1,749,047 ha in *Quercus spp.* The RA selected accounted for 33% of the number of cases registered and 40.1% of the wooded area burned in peninsular Spain in the 32 yr period considered. These percentages were quite variable in the nine woodland types (Table 2).

The total forest area proportion (shrublands and woodlands) was high in all the RA with median values up to 200,000 ha. The median woodland area was close to 150,000 and that of the woodland type for which the reference area was defined had median values close to 50,000 ha (Fig. 2). Mean elevation of the RA ranged from less than 400 m in *Eucalyptus spp.* to more than 1,000 m in *Pinus sylvestris*. Mean precipitation ranged from close to 500 mm in *P. halepensis* to more than 1,100 mm in *Eucalyptus spp.* Mean observed temperatures ranged from 10.4 °C in *P. sylvestris* to 14.4 °C in *P. pinea*. Differences in temperature were small between the observed data and the data estimated from climate models but the expected increase in mean yearly temperature in the climatic scenarios B2 and A2 was clear in all woodland types (Table 3).

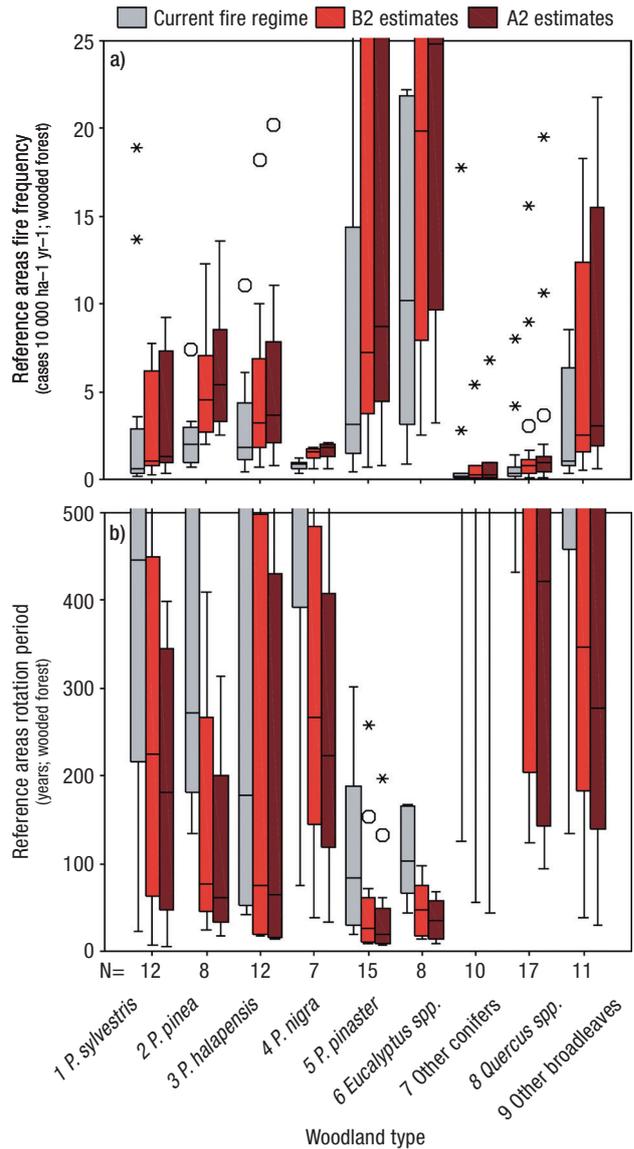
**Current fire regime (1974-2005) in the RA**

Based on the fire regime parameters calculated it was possible to distinguish three groups of woodland types (Fig. 1 and grey bars in Fig. 4a and b). The first group would be integrated by the two pines with lower fire incidence (*Pinus sylvestris* and *P. nigra*) and the three miscellaneous woodland types (“Other conifers”, *Quercus spp.* and “Other broadleaves”). Fire frequency was low (median values < 1 cases 10,000 ha<sup>-1</sup> yr<sup>-1</sup>) and the calculated fire rotation period was high (median values > 478 yr). The second group would be integrated by *P. pinea* and *P. halepensis* with median fire frequencies between 1.9 and 2 and median fire rotation periods between 272 and 178 yr respectively. Finally, the third group would be integrated by *P. pinaster* and *Eucalyptus spp.* the two woodland types with the higher fire incidence. Median fire frequency was 3.1 in *P. pinaster* and 10.2 in the *Eucalyptus spp.* Median calculated fire rotation period was 83 yr in *P. pinaster* and 103 yr in *Eucalyptus spp.* The spatial variability in the values registered for the fire rotation period within the RA defined for each woodland type was high for all of them. For some woodland types (e.g. *P. sylvestris* in the North West, *P. pinaster* in the West and *P. halepensis* in the East) the values of the rotation period (less than 50 yr) were extremely short (Fig. 1).

**Future fire regime estimates in the A2 and B2 climatic scenarios**

Future fire frequency estimates indicated an increase in fire frequency in all the RA defined for the nine woodland types. As an example, in the pine species with higher frequencies (*P. pinaster*) median values for the RA would increase from 3.1 cases 10,000 ha<sup>-1</sup> yr<sup>-1</sup> to 7.3 in the B2 and 8.7 in the A2 scenario (Fig. 4a).

Values of the fire rotation period higher than 500 yr were obtained in 52 of the 97 RA. This figure would decrease to 30 in the B2 and to 28 in the A2 climatic scenarios. The future estimates indicated shortenings of the rotation periods from median values of 572 yr to 267 yr in B2 and 222 yr in A2 for *P. nigra*, the pine species with a larger rotation period. The shorter rotation period was obtained for *P. pinaster* (median value of 83 yr), and this value would still decrease to 36 yr in the B2 and 20 yr in the A2 climatic scenarios. The longer rotation periods (much longer than 500 yr) were obtained for the “other conifers”, *Quercus spp.* and “other broadleaves”. In these three woodland types values shorter than 500 yr are expected in the A2 scenario for *Quercus spp.* (median value of 421 yr) and in

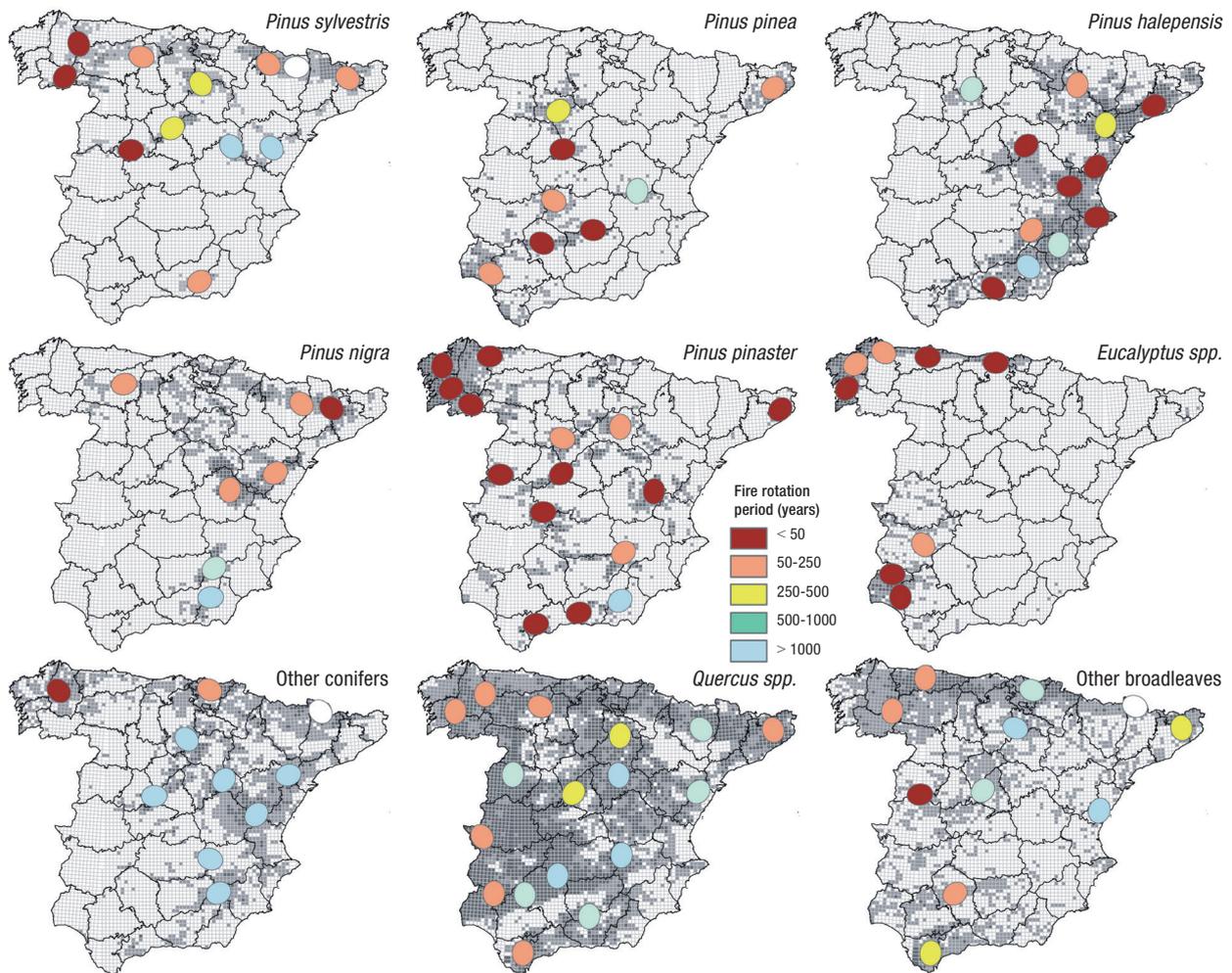


**Figure 4.** Box-plots showing the range in the calculated values of (a) the fire frequency and (b) the fire rotation period of the wooded area in the current (1974-2005) and estimates in the future climatic scenarios (SRES A2 and B2) for the RA defined for each of the nine woodland types. The number of reference areas selected for each woodland type is displayed along with some outliers (circles) and extreme (stars) values.

“other broadleaves” in the B2 and A2 with median values of 346 yr and 276 yr respectively (Fig. 4b).

In *Pinus sylvestris* woodlands, fire rotation periods shorter than 50 yr were registered in two RA located in north-western Spain (Fig. 1). Under the A2 scenario, one more RA located in the western Central Range was predicted to experience this short rotation period (Fig. 5). In *P. pinea* none of the RA displayed rotation periods shorter than 50 yr in the 32 yr considered but this fire rotation period value was estimated to occur in three out of the eight RA under the A2 scenario. In

## Estimated fire regime A2 scenario (2071-2100)



**Figure 5.** Maps displaying the location of the RA defined for the nine woodland types and the fire rotation period estimates for the A2 climatic scenario (2071-2100). In three of the 100 RA there are no estimates of the future fire rotation period (displayed in white). Grey tones indicate the proportion of area covered by the woodland type of concern in each 10 x 10 km grid-unit: light grey between 100 and 1,000 ha and dark grey more than 1,000 ha.

*P. halepensis*, the number of RA with shortest fire rotation periods (< 50 yr) would increase from three to six under the A2 scenario. In *P. nigra*, while none of the seven RA registered the shortest rotation period, one RA located in north-eastern Spain would experience lower than 50 yr rotation periods under the A2 scenario. In *P. pinaster*, values of the rotation period shorter than 50 yr were obtained in six of the 15 RA defined (Fig. 1). In the A2 scenario these values were estimated in 11 of the 15 RA. In the *Eucalyptus spp.* only in one of the eight RA the calculated value of the rotation period was shorter than 50 yr (Fig. 1) but this figure would increase to five RA in the A2 scenario (Fig. 5). In the “Other conifers” the calculated rotation periods were very long (longer than 1000 yr) in all the RA except in one located in north-western Spain in which the estimated rotation period would be shorter

than 50 yr in the A2 scenario (Fig. 5). In all the RA defined (17) for the *Quercus spp.* the rotation periods obtained were longer than 250 yr (Fig. 1) while it was estimated that in seven of them the rotation period would be shorter than 250 yr in the A2 scenario. Finally, in the “Other broadleaves” type only in one of the 11 RA defined the rotation period was shorter than 250 yr while these figure were estimated in four RA in the A2 scenario (Fig. 5).

## Discussion

### Recent fire regime in the woodland types

This work provides an evaluation of regional fire regime parameters focusing on individual forest species

and it also provides estimates for future climatic scenarios. Our results confirm a generally greater fire activity in pine woodlands, large differences between the different pine species and a large variability in the fire regime among RA belonging to the same woodland type. The results confirm also a higher fire activity in some of the species more used in afforestation and reforestation programs, such as *P. pinaster*, *P. halepensis* and *Eucalyptus spp.* These results agree with our starting hypothesis and point to a large relevance of the region in which the RA is located in addition to the woodland type. These patterns could be due to several factors such as the strong effect of human activities (Moreno *et al.*, 1998; Pausas *et al.*, 2008; Moreira *et al.*, 2011; Brotons *et al.*, 2013) and also to gradients of forest productivity which are very relevant in fuel accumulation and therefore in fire activity (Vázquez *et al.*, 2006) that agree also with the global fire-productivity relationship suggested recently by Pausas & Ribeiro (2013).

Our results on the current fire regime in pine woodlands show large differences between species. *Pinus sylvestris* and *P. nigra* registered a low fire incidence in most of the RA followed by *P. pinea* and *P. halepensis* with a generally medium incidence and finally a large incidence in the RA selected for *P. pinaster*. These three groups defined according to the fire regime registered partially agree with the strategies (or syndromes) proposed to understand the role of fire in the patterns of diversification of pines (Keeley, 2012). According to this work, *Pinus sylvestris*, *P. pinea* and *P. pinaster* should be considered as belonging to the fire-tolerating syndrome (able to survive low intensity surface fires due to several traits such as height coupled with the self-pruning of dead branches and a thick bark). On the other side *P. halepensis* would belong to the fire-embracer pine syndrome that enhance the spread of wildfires by limited self-pruning of dead branches that increase the probability of lethal crown fires (Keeley, 2012). However, other authors have already pointed out that the late flowering and absence of serotinous cones in *P. sylvestris*, *P. nigra* and *P. pinea* indicate that their natural forest did not evolve under conditions involving frequent crown fires while, by contrast, serotiny and early flowering in *P. halepensis* and *P. pinaster* make them able to regenerate successfully after the fire (Tapias *et al.*, 2004). Meaningfully, these last two species -the most widely used in afforestation in Spain- are those more affected by fires (33% and 24% respectively of the wooded area burned).

Although our calculations of the fire rotation period are based only in the wooded forest area, which accounts for just 37% of the total area burned in the 32

years considered, it is worthwhile to compare some fire rotation periods obtained in other places as a reference. In Portugal, the mean fire rotation period is estimated to be 82 years (1975-2005) while calculations in 21 forest regions ranged from 31 to 762 yr. In that period a 60% of the area affected burned only once, a 23% burned twice and a 17% three or more times (Oliveira *et al.*, 2012). Moreno *et al.* (1998) estimated that a fire rotation period shorter than 100 yr occurred in 22% of the grid-units in which peninsular Spain was divided during the period 1974–1994. A fire rotation period of 51 yr was reported for a study in a *P. pinaster* dominated area in central Spain while a spatially explicit calculation (taking into account the areas burned more than once) of this metric yielded 64 years (Vázquez & Moreno, 2001). Díaz-Delgado *et al.*, (2004) estimated a fire rotation period of 133 yr in Catalonia, NE Spain, for the period 1975-1998.

The variability detected in the RA belonging to the same woodland type has been large. For example, in *P. halepensis* the median value (in the 11 RA) of the fire rotation period is 182 yr but three of them have registered values shorter than 50 yr. Some of the largest fires registered in Spain have occurred on the Mediterranean coast and many of these were caused by lightning, as happened in 1994 (Moreno *et al.*, 1998). These fires affected to a large extent afforested or previously cultivated areas colonised by *P. halepensis* (e.g. Moya *et al.*, 2007). *P. halepensis* forests have been the most affected by high intensity crown fires (Eugenio *et al.*, 2006; Alvarez *et al.*, 2012). In a comparison between natural and planted pine forest in Spain, planted pine forests were shown to occur on poorer soils, to experience higher human-caused disturbance rates (fire frequency and human-caused mortality) and to have structural characteristics different from natural pine forests (Ruiz-Benito *et al.*, 2012). Species composition has also been demonstrated to affect deeply the probability of fire in NE Spain (González *et al.*, 2006). Focusing on other pines (e.g. *P. sylvestris* and *P. pinea*) it is clear that there are some RA with low fire incidence and long fire rotation periods and other RA with short rotation periods. In the case of *P. sylvestris* these last are located in north-western and central-western Spain, out of the current main distribution area of the species while in *P. pinea*, short rotation periods occur in South-Central Spain. In both cases, there has been an important spread through afforestation of the surfaces covered by these species.

At a regional scale is relevant to note also the strong relationship between fire activity and the productivity of the area. This pattern has been suggested for peninsular Spain (e.g. Vázquez *et al.*, 2006) and one possible explanation is that higher productivity is linked to more

favourable site conditions for vegetation growth that could give rise to larger fuel loads and faster biomass accumulation. These patterns could partially explain the high incidence and recurrence of fires in north-western of the Iberian peninsula: the Spanish region of Galicia (e.g. Vélez, 1990; Martínez-Fernández *et al.*, 2013) and also north of Portugal (e.g. Rego, 1991; Oliveira *et al.*, 2012). In any case, land-uses and human activities are also determinants in fire regimes in humanized landscapes.

### Fire regime estimates and forest changes

Our analyses assume a 3.7 times increase in the forest area burned in the RA defined in the A2 scenario. This figure is close to the 3.9 mean ratio provided by Vázquez *et al.* (2012) for the wooded area burned in the work used as a base for the analyses performed here. This 3.7 times increase is translated into shorter fire rotation periods which is in agreement with the results obtained in other approaches carried out in the Iberian Peninsula. In Portugal, Carvalho *et al.* (2010) reported even greater values than the ones used in this work with an increment of 478% (> 5.7 times the current level) for the SRES A2 scenario. A simulation analysis performed in three areas of NE Spain show that currently fuel-humidity limited regions could suffer a drastic shift of fire regime with an up to 8 times increase of annual burned area while the impact is predicted to be less pronounced in drier areas (Loepfe *et al.*, 2012).

Fire regimes are dynamic and vary in response to changes in drivers such as ignition frequency, vegetation patterns, climate and human activities (e.g. Bond & van Wilgen, 1996; Moritz *et al.*, 2012; Pausas & Fernández-Muñoz, 2012). In spite of the uncertainties associated with future climatic scenarios and the simplicity of the models, the results point to a significant increase in fire activity in all the RA defined. This would lead to a shortening of the fire rotation period that could affect negatively key processes such as regeneration (e.g. Carnicer *et al.*, 2014). The expected effects of the increasing fire activity could lead to changes in the composition of species and also to changes in forest structure, as already postulated by other authors (Mouillot *et al.*, 2002; Vilà-Cabrera *et al.*, 2012). *P. sylvestris*, *P. nigra* and *P. pinea* frequently exhibit problems for regeneration after the fire, probably due to the lack of an aerial seed bank (Tapias *et al.*, 2004) but other factors, including different eco-physiological needs should not be dismissed (Rodrigo *et al.*, 2004; 2007). Thus, our results support and run in parallel with

those that have predicted vegetation changes as a consequence of changing fire regimes (e.g. Rodrigo *et al.*, 2004; Schumacher & Bugmann, 2006).

Even for the pine species with easier regeneration after fire the short recurrence intervals are a problem for the sustainability of woodland. The recruitment of *P. halepensis* is limited by too short fire recurrence intervals (Eugenio *et al.*, 2006) and can be replaced by grasses or shrubs under recurrent fires (e.g. Arnan *et al.*, 2013). Oliveira *et al.* (2012) indicated that the recent fire regime observed in Portugal has already led to a substantial decrease in the area of *P. pinaster*. The different minimum maturation time (or age needed for the first effective reproduction) between species, but also between populations within species can be a key factor when dealing with a shorter fire recurrence period (Tapias *et al.*, 2004; Climent *et al.*, 2008). Also, genetic and plastic differences in other traits, like cone serotiny (in the fire-embracing strategy) or bark thickness (in the fire-avoiding strategy) can be critical in order to understand future individual survival and recruitment under altered fire regimes (e.g. Rigolot, 2004)

The fire regime estimates performed are only based on the effect that temperature increase will have on the empirical relationships established between fire regime and climate. That is, we have only considered the direct effects of climatic change on fire activity. However, the overall picture is more complex and includes the effects of climatic change on vegetation composition and structure and also the effects of vegetation changes on fire regimes (Mouillot *et al.*, 2002; Moritz *et al.*, 2012; Pausas & Ribeiro, 2013). The combination of both effects could make fire an even more important factor in vegetation dynamics, especially in fire prone areas. In the Mediterranean basin, predicted climate change is likely to exacerbate fire prone weather conditions, but the mid- to long-term impact of climate change on fire regime is not easily predictable (Loepfe *et al.*, 2012; Brotons *et al.*, 2013). These authors point to several key processes such as the possible reduction of biomass accumulation rates due to climate change and the role of humans on fuel management, fire ignitions and fire extinction.

The predicted increases in fire activity, illustrated by higher fire frequencies and the shortenings of the fire rotation periods shown in this work highlight the need to place more emphasis on fuel management and species use, the only direct aspects in which man plays a role. To minimize the adverse effects of fire on forest ecosystems they should be less susceptible to fire and more resilient to the expected future disruptions in fire regimes.

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