



Assessing the impact of prescribed burning on the growth of European pines



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ABSTRACT

Prescribed burning to reduce surface fuel loads could help preserve pine stands by increasing forest fire-resistance, but its effects on tree growth, especially growth of European pines, are poorly understood. Characterizing the short and mid-term effects of prescribed burning on *Pinus* growth could provide valuable input to inform fire and forest manager decision-making. Here we use dendrochronology and mixed modelling to investigate whether prescribed burning has differential effects on the mid-term growth of dominant and suppressed *Pinus* trees under different levels of fire severity via an approach comparing observed post-burning growth against inferred tree growth without burning. Results showed reduced growth of *Pinus halepensis* and suppressed *Pinus sylvestris* at year of prescribed burning. Mid-term post-burning growths were good for dominant *P. halepensis* and *Pinus nigra salzmannii* trees subjected to higher fire severities, whereas suppressed *P. nigra nigra*, *P. sylvestris* and *P. halepensis* grew less than expected without burning. Although prescribed burning tended to negatively affect the mid-term growth of *P. sylvestris* and *P. nigra nigra*, trees with higher pre-burn growth rates showed better post-burn recovery. The effect of fire severity on growth was positive for *P. nigra salzmannii* but negative for *P. nigra nigra*. These findings show that as time since burning elapses, growth recovery may depend on fire-tolerance of the pine species, degree of fire severity, tree characteristics and tree performance prior to prescribed burn. Understanding and balancing these factors in *Pinus* forests should help better plan prescribed burning, both in terms of desired fire intensity and required burning intervals, without altering tree vitality.

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1. Introduction

High-intensity fires threaten important ecological and social functions of *Pinus* species and their production value (Pausas et al., 2008). The negative impact of forest fires on the forest and its associated functions is expected to increase in the future, as model predictions point to increasing forest fire frequency, intensity and severity due to land-use and climate change (Flannigan et al., 2009).

Landscape-level fuel treatments strategically allocated in time and space can be combined with forest management efforts to reduce the extent and severity of forest fires, depending on vegetation type and historical fire regime (Agee et al., 2000). Fuel management in Europe traditionally relies on mechanical tools, but 10,000 ha yr⁻¹ of forest is currently being managed by prescribed burning (PB) in which planned fires are set and used by fire experts under mild weather conditions to meet a defined management

objective (Fernandes et al., 2013). PB is widely recognized in North America, South Africa and Australia, but it is still questioned in Europe although used marginally in Mediterranean countries like Portugal, Spain and France. *Pinus*-dominated stands cover 649,807 ha in Catalonia (NE Spain) without considering *Pinus uncinata* (Gracia et al., 2004), but only approximately 0.01% of the forest area was treated annually with PB (GRAF, 2011) whereas wildfires accounted annually for 0.8% of the total burned forest over the period 1999–2011 (González et al., 2007). The increase in number of large catastrophic fires in past decades (González and Pukkala, 2007) in Southern Europe has prompted the idea of establishing a less harmful fire regime, where the controlled spread of low-intensity unplanned fires is to be allowed and PB extensively applied as a cost-efficient way to reduce fuel continuities (Piñol et al., 2007; Regos et al., 2014). However, the requisite changes to the social, economic and legal restrictions limit the deployment of this new fire management policy for PB (Fernandes et al., 2013) and especially for unplanned fires (Regos et al., 2014). Besides, research of its potential effects on forest ecosystems and their accompanying services is still required, adding uncertainty.

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PB plans can lead into conflicts between protection, conservation and production goals, as PB may be beneficial for some ecosystem functions but indirectly affect others in an undesirable way (Driscoll et al., 2010). Post-PB tree vitality is a key aspect that warrants closer attention. PB may be perceived as aggravatory factor to the gradual process of tree death, both directly due to physical fire damage to the tree, and indirectly as a contributory factor exacerbating the effects of drought episodes and other stressors (Allen et al., 2010). In contrast, tree vitality can be partially affected, unaffected or even improved with time since burn, depending, among others, on the species, fire severity and tree size (Peterson et al., 1991; Valor et al., 2013). It is difficult to obtain a metric of tree vigour, as there are no direct measures to reflect it, but indicators such as crown transparency or tree growth, among others, are feasible proxies of tree vitality (Dobbertin, 2005). For instance, some dendroecological studies use tree growth as an indirect measure of tree vitality after a perturbation such as drought (Martínez-Vilalta et al., 2011; Linares et al., 2014) or defoliation (Linares et al., 2014) and in some cases even to predict tree mortality (Bigler and Bugmann, 2004). Tree vitality is diagnosed by comparing the stressed growth of a tree to a reference growth of the same tree (e.g. growth prior to the stress or modelled without stress) or the growth of other unstressed trees nearby (Dobbertin, 2005).

Postfire growth is essentially regulated by tree attributes (e.g. species, size, competition, age) and fire regimes (e.g. pattern, size, intensity, severity, season and recurrence) (Keyser et al., 2010). There is a consensus that specific characteristics of affected trees are major sources of variability in tree growth response to fire, yet few studies have actually characterized these variations. Reductions in tree growth result from altered photosynthetic processes due to the physical damage to tree tissues caused by the fire (Chambers et al., 1986), whereas increases in tree growth can be observed in unaffected trees or in trees healing over time since fire due to increased light and soil nutrient availability or reduced tree competition (Certini, 2005).

There is far less information on postfire tree growth in European species than species studied in North America and Australia. The few European studies on this subject have focused on growth responses in fire-resistant species (Agee, 1998) after low-severity forest fires in *Pinus nigra salzmannii* (Valor et al., 2013), *Pinus sylvestris* (Beghin et al., 2011; Blanck et al., 2013) and the fire-resistant fire-endurer (Fernandes et al., 2008) *Pinus canariensis* (Rozas et al., 2011), with rare studies on tree growth response to PB in *Pinus pinaster* (McCormick, 1976; Botelho et al., 1998). There has been little focus on fire-evader species (Agee, 1998) like *P. halepensis*, whether after forest fires (Battipaglia et al., 2014a) or after PB (Battipaglia et al., 2014b). Here, we studied the effects of PB as a surface fuel hazard reduction strategy on the short- and mid-term growth of pine species with contrasted tolerance to fire. Our aims were: (1) to ascertain whether there are consistent variations in post-PB growth (release or suppression) in *P. halepensis*, *Pinus nigra ssp. salzmannii*, *Pinus nigra ssp. nigra* and *P. sylvestris* subjected to PB and on sites with contrasting climatic conditions, and (2) to characterize individual tree growth response to PB over time since burning, based on crown status and fire severity as influencing variables. We hypothesize that low-intensity PB should be neutral or even improve tree vigour in more fire-resistant species like *P. nigra* and *P. sylvestris* but weaken tree vigour in the fire-evader species *P. halepensis*, but that these responses may change over time post-PB. The studied pine species are not only ecologically important in the region but also in terms of economic value as they represent annual cuttings of 350,000 m³, i.e. over 60% of the region's total harvested timber (Idescat, 2014) and subtain the yield of non-wood forest products as relevant in the region as mushrooms (Bonet et al., 2014). Determining the duration

and intensity of PB impact on growth and the subsequent recovery patterns (if any) should help refine PB timing and intensity.

2. Material and methods

2.1. Study sites

Study sites were obtained from the PB database developed by the Forest Actions Support Group of the Catalan Government (GRAF). From the database, we selected study sites treated with a PB at least 4 years prior to starting the study, (including year of PB) and dominated (>50% of basal area) by one of the main pines in the region (*Pinus halepensis* Mill., *Pinus nigra*, including two subspecies *Pinus nigra ssp. salzmannii* (Dunal) Franco and *Pinus nigra* Arnold. ssp. *nigra*, and *Pinus sylvestris* L.). A total of 14 study sites, distributed across the region of Catalonia (NE Spain), were selected for tree-ring sampling: 4 dominated by *P. halepensis*, 4 by *P. nigra ssp. salzmannii*, 3 by *P. nigra ssp. nigra* and 3 by *P. sylvestris* (Table 1 and Fig. 1).

P. halepensis study sites tend to be in areas of dry Mediterranean climate whereas *P. nigra salzmannii*, *P. sylvestris* and *P. nigra nigra* study sites fell into temperate cold sub-Mediterranean climate. Based on Spanish Meteorological Agency (AEMET) data, over 1986–2010 (Fig. 2), annual mean temperature and mean precipitation on the selected study sites were 14.3 °C and 491.4 mm for *P. halepensis*, 12.9 °C and 631 mm for *P. nigra salzmannii*, 11.8 °C and 589.3 mm for *P. nigra nigra*, and 11.1 °C and 534.9 mm for *P. sylvestris*.

2.2. Tree selection and tree-ring measurement

2.2.1. Tree selection

The criteria for cored-tree sampling aimed to select a minimum of 8 trees from each combination of crown status (dominant or suppressed) and fire severity class (high or low) at each study site. However, the target of 8 trees per combination was not always reached due to an absence of candidate trees in some of the grouping categories (Online Supplementary material).

Dominant trees were defined as trees whose crown width above the main canopy layer of the stand, intercepting direct sunlight, and among the larger of the stand in terms of stem diameter (Kraft, 1884). Suppressed trees were defined as trees whose crown remains entirely or partially below the main canopy, receiving none or little direct sunlight, and among the smaller of the stand in terms of tree diameter (Kraft, 1884). The classification of the trees on these two crown status categories was implemented to facilitate the analysis, requiring from the assumption that codominant trees were considered as a dominant and intermediate trees fell into the suppressed category. In order to define the fire severity classes (FSEV), the tree's bole char height (BCH) was measured as an indirect measure of cambium damage (Regelbrugge and Conard, 1993; Kobziar et al., 2006; Keeley, 2009). The degree of fire severity at the tree level can vary within a single PB depending mainly on micro-site fuel load and continuity, as meteorological conditions are rather constant. Trees were grouped into the high-severity class when $BCH > 1$ m and the low-severity class when $BCH < 0.5$ m. Therefore, both crown status (CS) and severity classes (FSEV) were defined as categorical variables with two levels of Dominant/Suppressed and High/Low, respectively. We assumed that CS at burn year was the same as in 2012 when we collected the data, because PB did not reduce tree density and no management operations were executed after the PB. Diameter at breast height (DBH) and total height (H_t) were measured in each tree (Table 2). In addition, the 5 trees closest to the target tree were measured for BCH, and their average BCH, including the measured

Table 1
Description of the PB sites.

Pine species	Study sites	Burn year	Burnt surface (ha)	Tree density (tree/ha)	Slope (%)	Aspect	Elev. (m.a.s.l)
<i>P. halepensis</i>	Ph1	2003	0.8	1089.4	24.0	E	340
	Ph2	2005	6.0	520.0	9.9	NW	261
	Ph3	2006	0.4	1645.6	22.0	NW	400
	Ph4	2007	5.8	980.8	20.0	NE	210
<i>P. nigra salzmannii</i>	Pns1	2003	2.2	836.1	34.6	N	670
	Pns2	2003	3.7	753.2	35.9	N	492
	Pns3	2003	0.9	1652.8	30.0	NE	737
	Pns4	2003	1.8	808.1	30.0	NE	730
<i>P. nigra nigra</i>	Pnn1	2002	9.5	830.7	27.7	NE	1049
	Pnn2	2003	3.4	1323.2	24.9	W	902
	Pnn3	2003	2.6	1031.5	10.0	NW	878
<i>P. sylvestris</i>	Ps1	2002	9.5	1425.5	30.5	N	1049
	Ps2	2002	5.2	256.1	10.0	N	1010
	Ps3	2003	0.9	708.4	30.0	NE	737

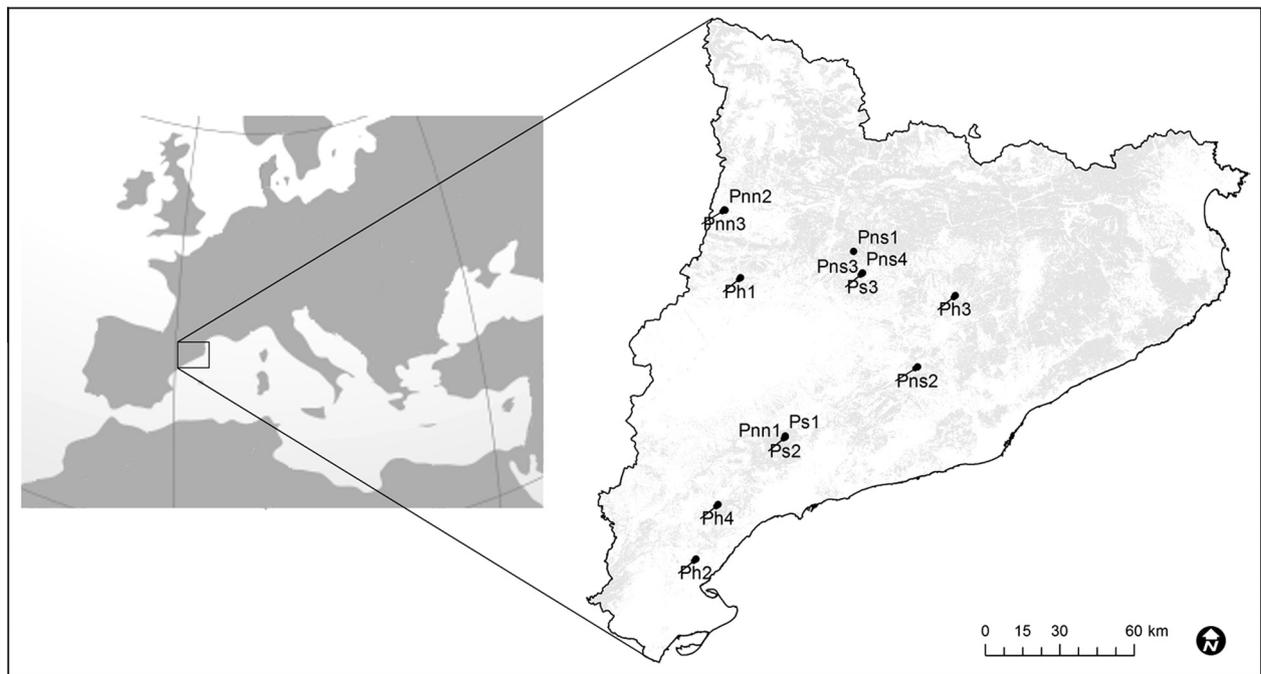


Fig. 1. Site distribution across Catalonia (NE Spain). Ph = *P. halepensis* sites, Pns = *P. nigra salzmannii* sites, Pnn = *P. nigra nigra* sites, Ps = *P. sylvestris* sites.

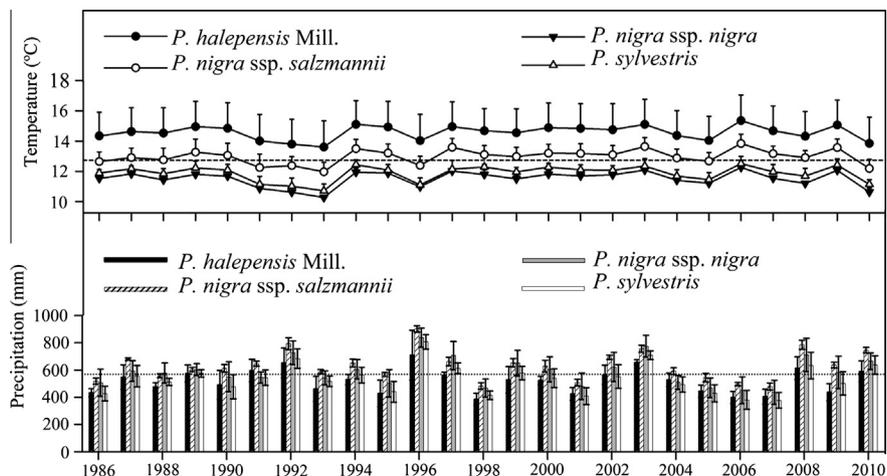


Fig. 2. Evolution in mean annual temperature and cumulative precipitation on the study sites, grouped by pine species. The standard deviation of the climatic variables among sites dominated by the same pine species is shown.

Table 2
Means \pm standard deviation of diameter at breast height (*DBH*), total height (*Ht*), bole char height (*BCH*) and local fire severity (*BCH_{local}*) at sampling year (2012) and burn-year diameter at breast height (*DBH_b*) and age (*AGE_b*), for each crown status–fire severity combination within each pine species.

Pine species	Crown status–severity	Sample size (n)	<i>DBH</i> (cm)	<i>DBH_b</i> (cm)	<i>Ht</i> (m)	<i>BCH</i> (m)	<i>BCH_{local}</i> (m)	<i>AGE_b</i> (years)
<i>P. halepensis</i>	Dominant-High	25	22.5 \pm 4.8	12.5 \pm 3.3	12.4 \pm 4.2	1.2 \pm 0.1	0.8 \pm 0.06	32.7 \pm 10.4
	Dominant-Low	30	19.7 \pm 3.6	12.6 \pm 2.5	12.8 \pm 5.8	0.3 \pm 0.1	0.4 \pm 0.04	36.6 \pm 20.0
	Suppressed-High	16	17.7 \pm 4.0	9.0 \pm 1.9	10.8 \pm 1.9	1.2 \pm 0.2	0.9 \pm 0.09	35.5 \pm 17.5
	Suppressed-Low	17	14.3 \pm 2.7	8.6 \pm 2.0	10.3 \pm 2.6	0.3 \pm 0.1	0.4 \pm 0.04	34.7 \pm 16.8
<i>P. nigra salzmannii</i>	Dominant-High	20	24.9 \pm 5.8	13.6 \pm 6.3	12.9 \pm 2.4	1.2 \pm 0.2	0.8 \pm 0.12	55.9 \pm 14.1
	Dominant-Low	20	20.1 \pm 5.0	10.4 \pm 3.0	12.9 \pm 2.1	0.3 \pm 0.1	0.4 \pm 0.06	51.6 \pm 18.2
	Suppressed-High	20	15.9 \pm 4.3	11.8 \pm 4.6	9.2 \pm 1.4	1.2 \pm 0.2	0.9 \pm 0.06	52.7 \pm 18.0
	Suppressed-Low	20	13.3 \pm 2.7	9.6 \pm 2.4	9.4 \pm 1.8	0.3 \pm 0.1	0.5 \pm 0.03	55.8 \pm 15.2
<i>P. nigra nigra</i>	Dominant-High	21	21.6 \pm 3.7	13.1 \pm 2.2	11.3 \pm 1.3	1.2 \pm 0.2	0.9 \pm 0.09	31.2 \pm 10.4
	Dominant-Low	20	21.1 \pm 5.4	12.7 \pm 3.5	10.6 \pm 1.4	0.2 \pm 0.1	0.4 \pm 0.06	32.7 \pm 10.7
	Suppressed-High	21	15.6 \pm 4.1	9.8 \pm 3.1	10.3 \pm 1.9	1.2 \pm 0.1	1.0 \pm 0.11	31.2 \pm 10.6
	Suppressed-Low	18	14.2 \pm 3.8	9.2 \pm 2.5	9.1 \pm 1.6	0.3 \pm 0.1	0.4 \pm 0.05	32.4 \pm 11.1
<i>P. sylvestris</i>	Dominant-High	20	23.9 \pm 3.7	16.3 \pm 3.5	12.7 \pm 1.9	1.2 \pm 0.2	0.7 \pm 0.11	46.2 \pm 10.4
	Dominant-Low	20	20.7 \pm 4.6	15.1 \pm 3.7	12.5 \pm 1.6	0.3 \pm 0.1	0.5 \pm 0.03	44.3 \pm 11.6
	Suppressed-High	20	15.9 \pm 3.8	12.0 \pm 4.0	10.0 \pm 1.9	1.2 \pm 0.3	0.9 \pm 0.39	42.3 \pm 11.4
	Suppressed-Low	20	15.4 \pm 4.7	9.8 \pm 3.7	10.2 \pm 1.8	0.3 \pm 0.1	0.7 \pm 0.34	40.1 \pm 13.2

target tree *BCH*, was calculated (*BCH_{local}*) and used as a proxy of the intensity of the fire once it reached our target tree (Table 2).

2.2.2. Tree-ring measurement

A total of 328 trees (88 *P. halepensis*; 80 *P. nigra salzmannii*; 80 *P. nigra nigra*; 80 *P. sylvestris*) were ultimately selected and cored to the pith during May and June 2012. Trees were cored with an increment borer at 1.30 m and as far away as possible from fire scars (Mutch and Swetnam, 1995) or in our case the most charred side of the stem, as scars were seldom visible. All extracted cores were prepared following standard dendrochronological techniques (Stokes and Smiley, 1968). Tree-rings were measured from year 2011 until pith, as late wood in spring 2012 was not still defined (Rinn, 1996), using a Lintab 3 measuring system coupled to TSAP tree-ring software (Frank Rinn, Heidelberg, Germany), with a precision of 0.001 mm. All cores were dated and visually cross-dated to detect the presence of false and incomplete rings. Within each of the crown status and fire severity combinations, cross-dating was validated using COFECHA software (Holmes, 1983), which calculates cross-correlations amongst individual series of tree growths. For each cored tree, the *DBH_b* and *AGE_b* were estimated by dendrochronological methods (Table 2). Mean interseries correlation (*Rbar*) was calculated to assess agreement between individual series in each combination (Online Supplementary material). The majority of the trees showed *Rbar* values above 0.3, except for some suppressed trees that still were kept for growth modelling as the correlation was positive (Martín-Benito et al., 2007; Martínez-Vilalta et al., 2011).

2.3. Climatic data

Monthly precipitation (*P*) and temperature (*T*) from 1985 to 2010 were downscaled for each study site from climatic records and projections elaborated by the Spanish Meteorological Agency for each study site (AEMET) (Ninyerola et al., 2000, 2007). Monthly potential evapotranspiration (*PET*) was estimated using the Thornthwaite, 1954 method. Cumulative values (1985–2010) of *P* and *PET* were calculated for three time windows ranging from June (*jun*), August (*aug*) and September (*sep*) of the year before the formation of the tree-ring (*t-1*) to September of the year of ring formation (*t*). For each time window, two classes of drought indices (*DRI*) were calculated: difference between *P* and *PET* (*P-PET*), and *P* to *PET* ratio (*P/PET*). In total, six drought indices were calculated for each study site: *P-PET_{jun(t-1)-sep(t)}*, *P-PET_{aug(t-1)-sep(t)}*, *P-PET_{sep(t-1)-sep(t)}*, *P/PET_{jun(t-1)-sep(t)}*, *P/PET_{aug(t-1)-sep(t)}* and *P/PET_{sep(t-1)-sep(t)}*. The

time intervals and their associated drought indexes were selected based on previous knowledge on the influence of climate on the radial growth of the studied pines (Gutiérrez, 1989; Martín-Benito et al., 2007; Sanchez-Salguero et al., 2012; Herrero et al., 2013).

2.4. Data analysis and growth modelling

The yearly radial increments were converted into yearly basal area increments (*BAI*) as initial predictor variable to quantify growth variation due to PB, using the equation $BAI = \pi (r_t^2 - r_{t-1}^2)$, where *r* is the tree radius and *t* is the year of tree-ring formation.

The approach for assessing growth variation due to PB encompassed two phases and was based on comparing observed *BAI* (*BAI_{obs}*) against inferred *BAI* (*BAI_{inf}*). First, pre-burning *BAI* was modelled using tree characteristics and climate conditions as predictors and *BAI_{obs}* prior to burning as predicted variable (from 1985 till one year prior to PB). The selected models were used to estimate *BAI_{inf}*, thus assuming that no PB ever happened. Afterwards, the *BAI_{inf}* was compared against the post-burning *BAI_{obs}* so as to assess the PB effect on annual variations in *BAI*. Therefore, if on a given year PB had a positive effect on growth, the *BAI_{obs}* of that year was supposed to be greater than the corresponding *BAI_{inf}*. On the other hand, if PB had a negative effect on tree growth, the *BAI_{obs}* was supposed to be smaller than the *BAI_{inf}*. Second, the standardized difference between *BAI_{obs}* and *BAI_{inf}* was used as a predicted variable (*BAI_{dif}*) to model the impact of PB on growth during the first four years after PB, using tree and fire severity variables.

2.4.1. Inferring growth under the no-burning scenario

For each pine species, a linear mixed model (Eq. (1)) was fitted for predicting *BAI* under a non-burning scenario using the best-fit drought index (*DRI*) (see Section 2.3) and breast height diameter at burnt year (*DBH_b*) as independent variables:

$$\ln(BAI_{k(jy)} + 1) = \beta_0 + \beta_1 \cdot f(DRI_{ky}) + \beta_2 \cdot DBH_b + \gamma_{0k(j)} + \alpha_{0k(j)} + \varepsilon_{k(jy)} \quad (1)$$

where β_0 is the overall intercept; β_1 and β_2 are the parameters adjusting the fixed effects, *k* is the index of the study site, *k(j)* is the index for tree nested in the study site, *y* is the index for year of measurement, γ_{0k} and $\alpha_{0k(j)}$ are the random effects associated with study site and tree, respectively, and $\varepsilon_{k(jy)}$ is the error term. Temporal autocorrelation of the error term was accounted for using an AR-(1) correlation structure. Models with the lowest Akaike

Information Criterion (AIC) (Burnham and Anderson, 2002) were selected and used, as mentioned, to infer the hypothetical growth of the no-burning scenario (BAI_{inf}), after applying a correction factor (Snowdon, 1991) to avoid bias due to log-transformation of the predicted variable. Annual BAI_{inf} was inferred until 2010, as growth in 2011 was excluded as 2011 climate data was unavailable. For each pine, the observed mean growth BAI_{obs} and mean BAI_{inf} was plotted against time-since-burning (TSB), which refers to number of years since application of the PB.

2.4.2. Modelling the impact of prescribed burning on tree growth

For each pine species, a linear mixed model was developed to describe short and mid-term impact of PB on growth. The predicted variable was the BAI_{dif} calculated as the difference between each of the first four years post-burning BAI_{obs} (including the year of PB) and the corresponding BAI_{inf} , divided by the square root of the pre-burning BAI_{obs} time series variance, in order to remove the effect of growth variability from the actual growth rate (Martínez-Vilalta et al., 2011). Models considered the following as potential fixed factors: time-since-burning (TSB), crown status (CS), fire severity class (FSEV) and local fire severity (BCH_{local}). Second interactions of FSEV with the other fixed factors except BCH_{local} were also included. All fixed factors and interaction terms were incorporated, and only the significant effects were retained. An AR(1) correlation structure was included to consider the temporal autocorrelation of the error terms. The non-independence of tree-ring growth data, as each tree growth responds to consecutive post-burning years, was resolved by adding a random intercept, which is a baseline for the multiple growth responses on the same individual tree. In order to characterize each tree baseline response to BAI_{dif} , random intercepts of each pine species model were regressed against age at burnt year (AGE_b) and the mean of the 5-year pre-burning BAI_{obs} (BAI_{obs5}). These two variables were considered potential factors explaining the random intercepts of the PB impact models of each pine. The residuals of all models developed presented no pattern, and highly correlated explanatory variables in the two phases of modelling were avoided. All analyses were performed using R software (v. 3.0.1, R Foundation for Statistical Computing) using the *nlme* package for linear mixed-effects modelling.

3. Results

3.1. Inferred growth under the no-burning scenario

For all pines, the best-fitted linear mixed models indicated that pre-burning BAI was significantly influenced by tree size at burning year and by the same drought month interval (Table 3).

The results showed that for all tree species, larger trees showed more growth than smaller ones, especially during years with no water restriction. The fixed part of each pre-burning BAI model

accounted for 0.47 of total variance in *P. halepensis* Mill., 0.11 in *P. nigra salzmannii*, 0.40 in *P. nigra nigra* and 0.40 in *P. sylvestris*.

Plotting the mean BAI_{inf} obtained using the fixed part of the models (Table 3) against the corresponding post-burning mean BAI_{obs} revealed that both followed a similar evolution within each pine (Fig. 3).

However, important variation appeared in specific years. For instance, at the year of PB, the BAI_{obs} was significantly lower than BAI_{inf} for *P. halepensis* and *P. sylvestris* (Fig. 3A and D) yet almost identical in the two *P. nigra* subsp. (Fig. 3B and C). The first year following the PB, there was a strong increase in BAI_{obs} compared to BAI_{inf} in *P. nigra salzmannii* (Fig. 3B). Lastly, the second year following the PB, there was a significant drop in BAI_{obs} compared to BAI_{inf} in *P. nigra nigra* which did not recover until year five after the fire, whereas *P. sylvestris* grew less than the predicted BAI_{inf} during the entire post-burning period (Fig. 3C and D).

3.2. Modelled impact of prescribed burning on tree growth

For all pines, growth variation was found to be significantly affected by TSB, with different signs depending on the year, and also by CS (except for *P. nigra nigra*), with dominant trees showing higher postburning growth (Table 4).

Severity of fire around the target tree, BCH_{local} , had a significant effect on the BAI_{dif} of *P. nigra* species: positive in *P. nigra salzmannii* and negative in *P. nigra nigra* (Table 4). None of the interactions was significant, and there were no differences between FSEV classes.

Evaluations of the short-term (TSB = 0) impact of PB on BAI_{dif} of *Pinus* species showed a negative BAI_{dif} for *P. halepensis*, *P. nigra nigra* with high BCH_{local} and suppressed *P. sylvestris* (Table 4). In contrast, PB was found to have a positive impact on BAI_{dif} in dominant *P. sylvestris* trees and all *P. nigra salzmannii* trees regardless of crown status, but in this case when BCH_{local} was high or extremely high (Table 4).

As time since PB elapsed (from TSB = 1 to TSB = 3), the impact of PB on the BAI_{dif} of dominant *P. halepensis* trees was found to be positive across the whole period, in contrast to suppressed trees that failed to show positive BAI_{dif} in any TSB year (Table 4). In the case of dominant *P. nigra salzmannii*, BAI_{dif} was also positive and increased in magnitude with increasing BCH_{local} , whereas suppressed *P. nigra salzmannii* showed negative BAI_{dif} values unless BCH_{local} was particularly high (Table 4). In the case of *P. nigra nigra* and *P. sylvestris* trees, PB had a negative impact on BAI_{dif} across the whole 3-year post PB period, regardless of tree status or fire severity (Table 4).

Finally, evaluation of the impact of random effects (AGE_b and BAI_{obs5}) on growth variation revealed that an increase in trees age had a negative effect on BAI_{dif} in *P. nigra nigra* and *P. sylvestris*, whereas an increase in pre-burning BAI_{obs5} had a positive effect on post-burn growth recovery in *P. nigra nigra* and *P. sylvestris* (Fig. 4) but no effect on the other two species.

Table 3

Best-fitted linear mixed models of the pre-burning basal area increment (BAI) from 1986 to the year before of the prescribed burning for each pine species.

Pine species	Best pre-burning BAI fitted model	R ²	AIC _{null} -AIC _{best}	Snowdon correction
<i>P. halepensis</i>	$-0.285 + 0.129 * DBH_b + 0.001 * P-PET_{jun(t-1)-sep(t)}$	0.64	494.6	1.11
<i>P. nigra salzmannii</i>	$0.552 + 0.027 * DBH_b + 0.001 * P-PET_{jun(t-1)-sep(t)}$	0.64	178.3	1.08
<i>P. nigra nigra</i>	$-0.431 + 0.122 * DBH_b + 0.398 * P/PET_{jun(t-1)-sep(t)}$	0.57	171.0	1.03
<i>P. sylvestris</i>	$-0.482 + 0.109 * DBH_b + 0.197 * P/PET_{jun(t-1)-sep(t)}$	0.55	133.1	1.08

DBH_b , diameter at breast height at burnt year.

$P-PET_{jun(t-1)-sep(t)}$, drought index calculated as the difference between precipitation and potential evapotranspiration from June before tree-ring formation to September of the year of tree-ring formation.

$P/PET_{jun(t-1)-sep(t)}$, drought index calculated as the ratio between precipitation and potential evapotranspiration from June before tree-ring formation to September of the year of tree-ring formation.

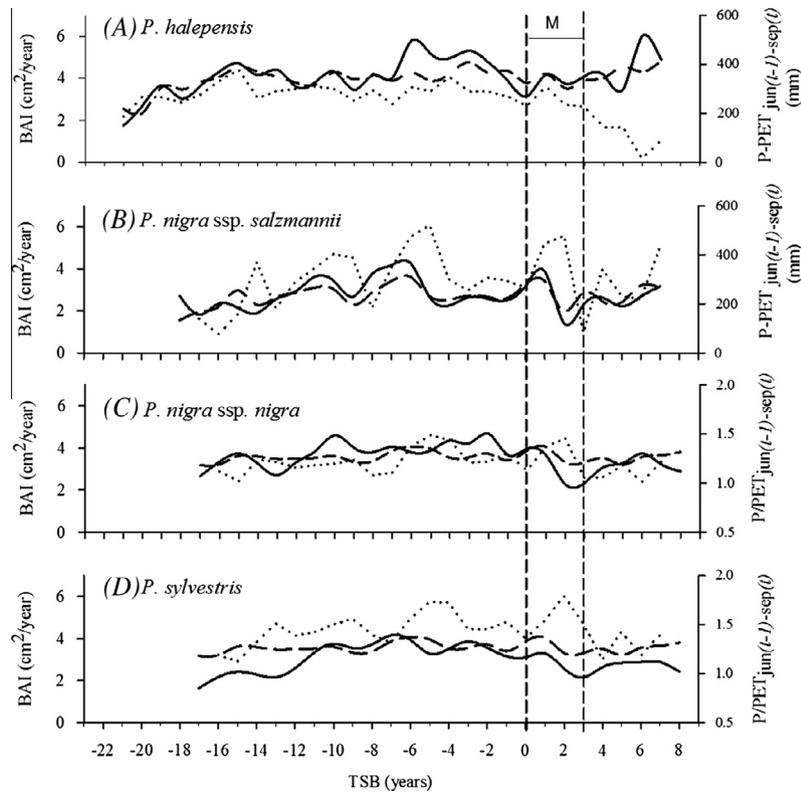


Fig. 3. Comparison of time-course trends of mean observed basal area increment (BAI_{obs}) (solid lines), inferred basal area increment (BAI_{inf}) in the case of no PB application (dashed lines), and the influencing drought indices (dotted lines) against time-since-burning (TSB) for each pine species. PB year (dashed vertical line) and M, modelling period of the PB impact on growth (see Section 3.2) are also shown.

Table 4
Summary of the models characterizing the impact of prescribed burning on growth variation (BAI_{diff}) in pine species.

Fixed effect	<i>P. halepensis</i>			<i>P. nigra salzmannii</i>			<i>P. nigra nigra</i>			<i>P. sylvestris</i>		
	Estimate	SE	P value	Estimate	SE	p-value	Estimate	SE	p-value	Estimate	SE	p-value
Intercept	-1.035	0.208	<0.001	-1.185	0.237	<0.001	0.572	0.253	<0.05	-0.551	0.188	<0.01
TSB = 1	0.374	0.117	<0.01	0.332	0.123	<0.01	-0.549	0.153	<0.001	-0.209	0.108	0.055
TSB = 2	0.567	0.128	<0.001	-0.132	0.133	0.324	-0.840	0.159	<0.001	-0.394	0.104	<0.001
TSB = 3	0.422	0.130	<0.01	-0.335	0.135	<0.05	-0.811	0.160	<0.001	-0.427	0.104	<0.001
CS = Dominant	0.693	0.246	<0.01	0.749	0.169	<0.001	0.087	0.221	0.694	0.608	0.250	<0.05
BCH_{local}	0.364	0.489	0.550	0.639	0.262	<0.05	-0.838	0.279	<0.01	-0.083	0.268	0.757

TSB, time since burning; CS, crown status; BCH_{local} , local fire severity.

Bold characters indicate significant effects ($p < 0.5$).

Model $R^2 = 0.71$ for *P. halepensis* (AIC = 1040.5), $R^2 = 0.52$ for *P. nigra salzmannii* (AIC = 897.7), $R^2 = 0.56$ for *P. nigra nigra* (AIC = 1031.1) and $R^2 = 0.79$ for *P. sylvestris* (AIC = 871.4).

Fixed Model $R^2 = 0.08$ for *P. halepensis*, $R^2 = 0.17$ for *P. nigra salzmannii*, $R^2 = 0.11$ for *P. nigra nigra* and $R^2 = 0.07$ for *P. sylvestris*.

4. Discussion

This study was designed to assess the impact of PB on the growth of European pines. The information generated here provides a framework to establish species-specific PB recommendations on timing and intensity designed to reduce the potentially negative impacts of PB on tree vitality.

Our results show that when the impact of PB on tree growth is assessed at the year of the PB its effect varies among pines (Fig. 3). Immediately after the PB, growth variations can be interpreted in the context of the ecological fire adaptations of the studied *Pinus* species. Especially those related with potential differences in bark thickness that may confer distinctive degrees of fire tissue protection among them particularly when young (Fernandes et al., 2012). Firstly, growths remained unaffected in fire-resistant species such as *P. nigra salzmannii* (Fig. 3B), probably, due to its pyro-resistant

morphological traits conferring fire tissues protection (e.g. thicker bark) (Fule et al., 2008) and to a lesser extent in *P. nigra nigra* (Fig. 3C). Although there is some evidence that *P. halepensis*, when adult, can withstand surface fires (Fournier et al., 2013), generally it is assumed that as a fire evader species its traits are less favourable to cope with low-intensity fires (e.g. thinner bark) (Keeley, 2012) explaining probably the observed reduction in growth immediately after the fire (Fig. 3A). Lastly, the growth reduction suffered by the moderately fire resistant *P. sylvestris* may be related to cambium damages caused by the fire due to its thinner bark when compared with *P. nigra* (Fig. 3D) (Tapias et al., 2004; Fernandes et al., 2008).

The findings of the study also highlight the importance of modelling tree growth response to PB as a function of tree and fire attributes to gain insight into individual tree responses (Table 4), as our results point to variations in the above-mentioned trends not just

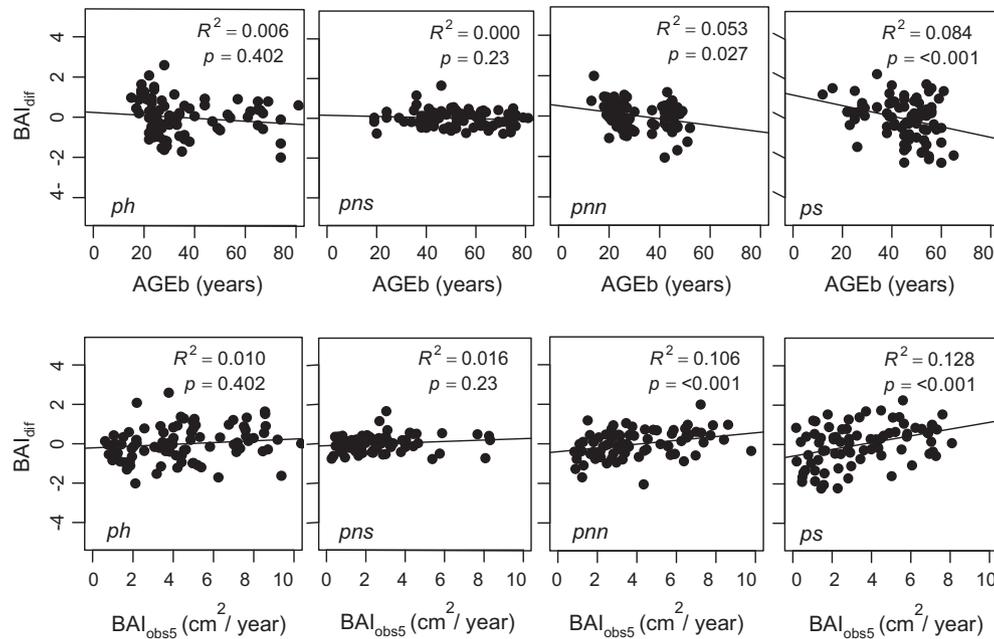


Fig. 4. Adjusted R^2 of tree age at burnt year (AGE_b) and mean of the 5-year pre-burning BAI (BAI_{obs5}) on growth variation (BAI_{diff}) for all pine species. The regression lines were estimated by fitting a linear model. $Ph = P. halepensis$, $Pns = P. nigra salzmannii$, $Pnn = P. nigra nigra$, and $Ps = P. sylvestris$.

at the year of PB but also as time increases. For example, as expected, crown status plays a major role in defining the immediate impact of PB on tree growth (Sutherland et al., 1991), with dominant trees suffering less growth reduction immediately after PB than suppressed trees. The reasons for this impact is presumably that smaller trees are more exposed to cambial injuries due to their thinner bark, especially at higher stem heights (Fernandes et al., 2012). The fact that *P. nigra nigra* was used for reforestation purposes, with homogeneous tree development, may be the cause of the absence of differences between crown status. Bark thickness, together with the temperature and the residence time of the fire beneath the tree, determine ultimately the degree of cambium damage (Bova and Dickinson, 2005). However, the fire severity indicator, BCH , used to create $FSEV$ classes was not found to have a significant effect on post-burning growth. The lack of a measurable impact of fire severity on growth could be due to the relatively small variability between the defined $FSEV$ classes (Table 2). With this regard the short-term reduction in growth observed in dominant *P. halepensis* may be explained by longer duration of burning at the base of the trunk due to their shorter needles present in the litter (Fonda, 2001) but also for its greater crown vulnerability to fire as it has the lowest canopy base height among Mediterranean pine species (Mitsopoulos and Dimitrakopoulos, 2007). In this sense the reduced growth in suppressed trees could be also due to the combined effects of fire damages, not only on the cambium, but, also, on the crown due to their lower canopy base heights, which probably allows flames to scorch a larger portion of the tree crown foliage, leading to higher rates of transpiration and inefficient photosynthesis bringing about a reduction in stem growth (Keyser et al., 2010). The amount of canopy scorched, if any, would have been an appropriate indicator for defining fire severity at the crown level (Fernandes et al., 2008), but the retrospective nature of our study impeded the measurement of crown scorch. Indeed, we did not find a significant interaction between $FSEV$ and CS that could explain, partially, potential fire crown damages and its effects on growth. However, when we looked at local fire intensity on our target trees, as expressed by BCH_{local} , we found a positive effect of BCH_{local} on the growth of *P. nigra salzmannii*, in agreement

with a previous study by Valor et al. (2013). A possible explanation for this increased growth could be that the average BCH on a small area can provide an approximation of flame length and fire intensity, with higher intensities translating to higher soil organic matter mineralization and therefore higher post-fire soil fertility (Pausas et al., 2003), although the effect of duff moisture may disrupt this relationship. This potential effect of enhanced nutrient availability has been observed to last from 1 month to 5 years depending on type of nutrient, burnt tree species, soil properties, burn intensity, and pathway of leaching processes (Certini, 2005). Conversely, the effect of BCH_{local} on growth was significantly negative on *P. nigra nigra*, highlighting the possibility of developing different adaptive characteristics depending on species origin and provenances. Thus, *P. nigra nigra*, a species whose origin has been traced to central Europe (Keyling et al., 2012), may be more proximately adapted to cope with perturbations such as snow or wind (e.g. open crown) rather than fire. In the case of *P. halepensis* and *P. sylvestris*, BCH_{local} had no significant effect on tree growth, which means there may be a trade-off between the effects driven by fire damage on cambium tissues and partially on the crowns, on these more fire-sensitive species, and the presence of fire-induced fertilization and removal of understory in the years following PB (Wyant et al., 1983). In contrast to *P. halepensis* and *P. nigra salzmannii*, which showed a tendency towards increasing post-burn growth, *P. sylvestris* and *P. nigra nigra* showed an intensified decrease in growth, similar to results reported on *P. sylvestris* during the first 5 years following fire (Beghin et al., 2011). Although these reductions in growth may be related to fire causing tissue damage, this growth variation might also be explained by the effect of cumulative drought events on post-PB growth (e.g. 2005, 2006 and 2007 (Fig. 2)) that were not captured in the pre-burn models, resulting in an overestimation of BAI_{inf} in *P. sylvestris* and *P. nigra nigra* which are particularly sensitive to drought events in this region. Nonetheless, it is also remarkable that in these species a better pre-PB growth performance resulted in a faster post-PB recovery (Fig. 4). Additionally, the age of trees was found to have a negative or none-significant effect on growth, depending on the tree species. However, it has to be mentioned that as age most

probably correlates to the dbh of the trees, a variable used to estimate the BAI_{inf} , and subsequently the BAI_{dif} , interpreting the impact of age on the variation of post-fire growth is difficult.

The models presented could be improved by considering fire intensity (e.g. residence time above a lethal temperature) or additional fire severity variables (e.g. crown scorch, duff consumed), which can be only obtained during or immediately after PB, but again, the retrospective nature of the study made this impossible to study. It should also be noted that the climatic dataset used, which is combination of records and projections using spatial interpolation tools could not reflect exactly the actual climate of the study sites. Note that on specific years, we used a lower number of tree-ring series to calculate mean BAI , as BAI was referenced to TSB and not to actual year growth, thus explaining the BAI_{obs} departures observed in *P. halepensis* from year 5 to year 8 (Fig. 3A). Future research should look closer at the short-term mechanisms and physical and biological processes liable to shape the mid-term effects of PB on tree vitality, and also explore the effects of recurrent PB on growth.

5. Conclusions

This study brings valuable information for fire and forest management decision-makers. It shows that the impact of PB on tree growth varies as time increases, and that this response is modified by the tree and fire characteristics of the pines. Our findings show that for all pines analysed, immediate post-PB growth was reduced in *P. halepensis* and *P. sylvestris* but remained unaffected in *P. nigra* spp. As time since burning increased, *P. halepensis* growth recovered and *P. nigra salzmannii* growth increased whereas *P. sylvestris* and *P. nigra nigra* growth decreased continuously. As a rule, dominant trees grew better than suppressed trees, indicating that larger trees have greater protection against fire probably due to its thicker bark and greater crown base height, even if suppressed trees grew less *per se* than dominant trees. Interestingly, higher fire intensities around the target trees appeared to enhance the growth of *P. nigra salzmannii* and to a lesser extent *P. halepensis*, probably due to fire-induced soil fertilization. Indeed, dominant *P. nigra salzmannii* and *P. halepensis* doubled diameter in 7–10 years after the PB (Table 2), being such increments observed in the region particularly at fertile sites (Trasobares et al., 2004a,b). From the presented results, it can also be stated that low intensity fires, as the ones promoted through PB, have a small impact on the short term growth of pine forests, and its use as a silvicultural tool has to be framed on the context of fuel reduction activities. However, it has to be mentioned that as fire intensity increases and post-fire time elapses significant and even positive increments on tree growth can be expected (Valor et al., 2013) if an adequate balance between competition reduction and the tissue damage caused to remnant trees is accomplished, offering a new window of opportunity for the use of PB as a forest management tool.

In summary, growth recovery may depend on the species morphological traits (e.g. tree size, crown status, bark thickness), the severity and intensity of the PB, tree performance prior to PB, the effects of tree healing, and increased light and soil nutrient availability postfire. Understanding and balancing these factors in *Pinus* forests should help forest and fire managers better plan PB operations, both in terms of desired fire intensity and required burning intervals.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.foreco.2015.02.002>.

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