Forest Ecology and Management 343 (2015) 101–109

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

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

Teresa Valor *, José Ramón González-Olabarria, Míriam Piqué

Forest Sciences Centre of Catalonia (CEMFOR-CTFC), Carretera de Sant Llorenç de Morunys, km2, 25280 Solsona (Lleida), Spain

ARTICLE INFO

Article history:
Received 23 October 2014
Received in revised form 10 January 2015
Accepted 3 February 2015
Available online 21 February 2015

Keywords:
Prescribed fire
Dendrochronology
Tree growth
Pinus halepensis
Pinus nigra
Pinus sylvestris

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 towards 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.

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.

* Corresponding author. Tel.: +34 653 591 552.
E-mail address: teresa.valor@ctfc.es (T. Valor).

http://dx.doi.org/10.1016/j.foreco.2015.02.002
0378-1127/

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 m^3, 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 (*Ht*) 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