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Effect of burnt wood removal on the natural regeneration of *Pinus halepensis* after fire in a pine forest in Tus valley (SE Spain)

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Abstract

To determine the effect of burnt tree removal on post-fire natural regeneration of *Pinus halepensis*, two 2 500 m^2 areas were selected six months after the fire in a totally destroyed mature (>70 years) pine forest. In one area, all the trees were cut down and removed 10 months after the fire and, in the other, all the trees were left standing (control). In each area, 20 permanent plots of 20 m^2 each were randomly placed, and all seedlings emerging within them labeled by individual numbered plastic tags. Emergence, mortality, density and growth (height) of 6649 *P. halepensis* seedlings were monitored during the first four post-fire years. Seedling emergence was concentrated in the first post-fire autumn–winter period. No positive effect on seedling emergence was detected as a consequence of burnt trunk dragging and subsequent turning over of soil. Wood removal produced an immediate average seedling mortality of 33%, and notably increased seedling mortality during the subsequent summer, probably due to increased exposure of seedlings to sunlight and the possible debilitation of many individuals by mechanical contact during burnt wood removal. A negative correlation of pine seedling mortality with height was detected, which increased significantly on wood removal in the third post-fire year. That is, short seedlings (<10 cm) in treated plots were the most likely to die during this period. In spite of the detrimental effect of wood removal on sapling survival, seedling density four years after fire in the cleared area was large (3.3 seedlings/m^2). Wood removal treatment reduced seedling growth: seedling height was significantly higher in the control stand, and differences in seedling height growth rate became particularly noticeable in the fourth post-fire year. The results denote that traditional wood removal practices do not threaten natural post-fire *P. halepensis* re-establishment if initial seedling density is large enough. However, further studies focused on wood removal effects on the final tree development level and other ecological aspects are necessary to choose adequate post-fire forest management.

Keywords: Tree felling; Mediterranean pine forest; Post-fire forest management; Seedling growth; Seedling recruitment

1. Introduction

Fire is an ecological factor which has played a key role in shaping plant communities of Mediterranean areas (Naveh, 1975; Trabaud, 1980; Pons and Thinon, 1987). The importance of this disturbance throughout the natural history of these communities is evidenced by the development of a set of adaptative mechanisms which confer on plant populations an active response capability to fire (Trabaud, 1987; Naveh, 1989).

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In the Iberian Peninsula, the burnt-forest surface has drastically increased during the last decades due to the accumulation of plant fuel in woodlands as a consequence of the gradual reduction of rural human populations, and the greater incidence of human intervention among main causes of wildfires. About 75 000 man-induced fires took place during the period 1986–1995. In 1994 alone, 3 65 371 ha of forest were destroyed by fire in Spain. Wildfires in this year were characterised by being especially intense and having long duration in a large number of cases. In the Tus Valley (Albacete Province, SE Spain), a man-induced wildfire lasted seven days and burnt a surface area of ca. 14 000 ha, mainly affecting *Pinus halepensis* Miller forests. This species is one of the most important forest species in the southern half of Spain, forming 40% of the tree-covered surface in the province of Albacete (Ortuño and Ceballos, 1977).

In spite of the abundant literature produced in the last years concerning *P. halepensis* response to fire in the Mediterranean Basin (Abbas et al., 1984; Trabaud et al., 1985; Moravec, 1990; Ne'eman et al., 1993, 1995; Saracino and Leone, 1993; Saracino et al., 1993; Thanos et al., 1996), and more recently in the southern Iberian Peninsula (May, 1992; Martínez-Sánchez et al., 1995, 1996; Herranz et al., 1997), there are very few studies which have tried to determine effects of different management practices in recently burnt forests on the post-fire natural recovery of this species (May, 1992; Ne'eman et al., 1993; Saracino and Leone, 1993).

Felling and removal of dead trees after fire for timber use is a traditional and widespread practice in Spain in most regions in the Mediterranean Basin. Apart from the commercial worth of trunks (i.e. 5 50 000 m^3 of wood from the burnt area in Tus Valley was sold after fire occurred in 1994), two reasons have been frequently presented by forest managers justifying this action:

1. Turning over of soil by trunk dragging during wood removal operations facilitates germination of pines, because contact between seeds which survived fire and mineral soil is favoured; and
2. removal of dead wood reduces the risk of xylophagous arthropod proliferation, which could negatively affect new pine development and survival in the future.

Some studies, however, have shown this practice as negatively affecting post-fire establishment of pine seedlings, since the slight cover provided by mature dead trees reduces insolation and, thus, mortality of seedlings during summer (Ne'eman et al., 1993; Saracino and Leone, 1993).

The aim of our study is to determine the effect of traditional management practices of tree salvage after fire on the establishment and development of *P. halepensis* seedlings during the early stages of post-fire plant recovery, testing the validity of the following two null hypotheses:

1. removal of wood favours pine seedling emergence through trunk dragging; and
2. remaining standing and intact burnt trees favour the growth of seedlings during the first post-fire years because of greater effects of shading.

2. Study area

The study area was located in the Tus Valley, in the SW of Albacete province (38°22′ N, 2°24′ W). As already mentioned, in August 1994, a man-induced wildfire, which started at different points in the centre of the valley, spread quickly and destroyed the vegetation in a surface area of 14 000 ha, mainly affecting forests of *P. halepensis* and, to a lesser extent, *P. pinaster* Aiton. The study was carried out in a burnt homogeneous mature stand (>70 years) of *P. halepensis*, located on a hillside with a slight slope (ca. 5%), sunny exposure. Density of pine trees before the fire was 425–450 trees ha^{-1}. Tree height ranged from 12 to 20 m, and canopy cover was 50–60%. Substratum in the area was dominated by carbonate materials, mainly limestone and dolomite (Rodríguez-Estrella, 1979). This substratum has originated soils consisting of calcisols, because of calcium carbonate leaching in the upper horizons. Altitude was 900 m and the climate was typically Mediterranean, with annual rainfall ca. 550 mm and an average annual temperature of 13°C. Under the dense *P. halepensis* canopy was a shrub layer with a total cover of ca. 45%, and where *Cistus monspeliensis* L., *C. albidus* L., *Rosmarinus officinalis* L., *Quercus ilex* subsp. *rotundifolia* Lam., *Juniperus oxycedrus* L., *Pistacea lentiscus* L., and *Daphne gnidium* L. dominated. In the selected stand, fire destroyed and totally consumed above-ground biomass of the understorey, and no pines survived. However, trunks were only superficially burnt and crowns slightly affected due to the large diameter and heights of trees so that they remained standing after the fire.

3. Methods

3.1. Experimental design

Two zones of 2500 m^2 at a separation of 50 m were marked in the chosen stand. All burnt pines in one of these zones were cut down and removed (cleared area) 10 months after the fire (June 1995), which is the normal time interval between fire and the beginning of wood removal operations in the region. In the other zone, pines were left standing (control area). Wood removal in the experiment followed the exact method traditionally used in the area: trees were cut by mechanical chain saws; trunks were dragged to the outside of plots by animal traction (mules), and finally, smaller twigs were removed from the cleared area manually. After this treatment, both these areas were fenced to prevent livestock and/or wild animals interfering in the plots.

In order to study pine seedling population dynamics, a set of 20 permanent plots of 4×5 m^2 were randomly placed in each area (control and cleared) six months after the fire. Nine samplings were carried out during the first three post-fire years. The first sampling was made six months after the fire (February 1995), when the experimental mechanism was set-up. The other samplings were carried out just before, and just after, the date of clearing (June 1995). The six other samplings took place as follows: October 1995, January 1996, June 1996, January 1997, June 1997, and June 1998. In the first sampling, all emergent *P. halepensis* seedlings were labelled by individual numbered plastic tags. In subsequent samplings, the number of dead seedlings which had been previously labelled was recorded and newly emerged seedlings were labelled, for assessing the rates of seedling mortality and emergence throughout the study period, respectively (Herranz et al., 1997). Seedling height was monitored from sampling in January 1996 (when most of the seedlings had reached 5–6 cm) to June 1998.

3.2. Data analysis

General suggestions in Zar (1984) were taken into account in the performance of statistics. Mortality, density, height, and growth of seedlings were compared between trees in control and cleared plots by Student's *t*-tests at the 0.05 significance level. Weighted means from plots in each stand were used for height and growth comparisons between treatments. In all the cases, normality and homoscedasticity were previously checked by the Kolmogorov–Smirnov and Cochran's tests, respectively. Mortality and growth data (in percentage) were submitted to an arcsine transformation for normal distribution adjustment.

The dependence of seedling survival on wood removal and seedling height was tested by logistic regression analysis. This statistical technique allows a prediction of a dichotomous (only two possible values) dependent variable from a set of independent nominal or numerical variables. The logistic regression model is expressed by the formula

$$p = 1/(1 + e^{-z})$$

where

$$z = a + \Sigma b_i x_i$$

and a is the constant for the non-linear model, b_i the coefficient estimated from data for the independent variable x_i, and p the probability of an event (dependent variable) to occur. The model was built by a forward stepwise variable selection using the Wald statistic (standard error values were low enough to permit its use) and the residual χ^2 statistic for determining variables to be included in or removed from the model, respectively. The nominal independent variable (wood removal: cleared or control) was recoded into dummy variables to represent every category.

4. Results

4.1. Seedling emergence

Most of the seedlings emerged during the first six post-fire months (first autumn–winter after fire). During the second post-fire autumn, another seedling

Table 1
Average emergence (in percentage; ±SD) of *P. halepensis* seedlings throughout the study period. (b) and (a) denote samplings immediately before, and after, wood removal treatment

Sampling date	Emergence rates	
	control	clear
Feb 1995	95.16 ± 2.64	93.74 ± 3.13
Jun 1995 (b)	1.64 ± 1.70	2.36 ± 2.35
Jun 1995 (a)	—	0
Oct 1995	0.28 ± 0.48	0.07 ± 0.21
Jan 1996	2.62 ± 1.98	3.25 ± 1.55
Jun 1996	0.31 ± 0.69	0.55 ± 0.64
Jan 1997	0	0
Jun 1997	0	0
Jun 1998	0	0

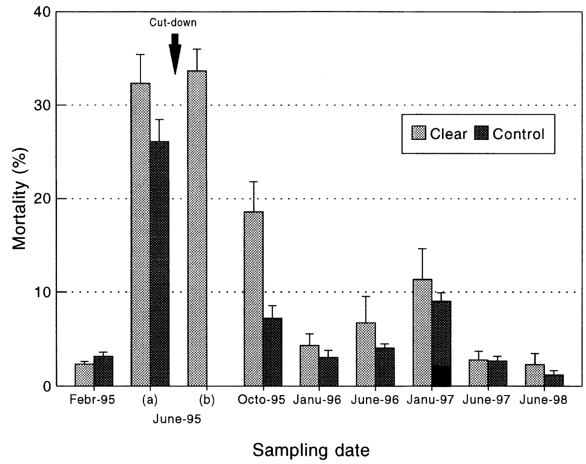

Fig. 1. Average mortality (in percentage; +SE) throughout the study period of *P. halepensis* seedlings which emerged during the first six months after fire. Abscissa: measurements immediately before, and after, wood extraction are labelled (b) and (a), respectively. The dark portion in the control bar corresponding to January 1997 represents mortality due to dead tree windthrow. Comparisons of mortality between cleared and control plots: February 1995: $t_{38} = 1.51$, n.s.; June 1995(b): $t_{38} = 1.61$, n.s.; June 1995(a): $t_{38} = 3.24^{**}$; October 1995: $t_{38} = 0.87$, n.s.; January 1996: $t_{38} = 0.92$, n.s.; June 1996: $t_{38} = 1.58$, n.s.; January 1997: $t_{38} = 1.58$, n.s.; June 1997: $t_{38} = 0.31$, n.s.; June 1998: $t_{38} = 0.81$, n.s.

emergence peak was recorded, but it was much less intense than the first one. From the third autumn after the fire, no pine seedlings emerged in either area (Table 1). Seedling emergence was not increased by wood removal. The pattern recorded in cleared plots was exactly the same as that in the control area (Table 1).

4.2. Mortality and density of seedlings

Seedlings which emerged during the first autumn after fire suffered very high mortality during the first post-fire spring: mortality ranged from 26% to 32%, even exceeded summer mortality (Fig. 1). Wood removal produced an immediate average mortality of 33%, and significantly increased seedling mortality during the subsequent summer in cleared plots ($t_{38} = 3.24$; $p < 0.001$; Fig. 1). Thus, in October 1995 (15 months after fire) cumulative average mortality in cleared plots was nearly twice as high as in control plots (62.13% and 32.88%, respectively). However, from this time, and in subsequent samplings, no significant difference in seedling mortality between both the treatments was recorded. Average mortality gradually decreased during the study, reaching values <3% from the third post-fire summer (Fig. 1).

From the second post-fire winter, standing burnt trees in the control area were windthrown, which produced an average seedling mortality of 3.08% at the end of the study period (Fig. 1).

Pine seedling mortality was negatively correlated with height, both in cleared and in control areas (Fig. 2(a–d)). Seedling survival probability was independent of treatment (clearing or not), except during the third post-fire year (June 1996–June 1997). However only short height classes were affected (Fig. 2(b and c)), so that small individuals experienced significantly greater mortality in cleared rather than in control plots during this period. In both these areas, survival probability of seedlings, reaching a height of 12–15 cm or taller, was very high from the second post-fire summer (95–99%; Fig. 2(b–d)).

The few seedlings which emerged between February 1995 and June 1998 (Table 1) had a final cumulative mortality of ca. 60% (57.63% in control plots, and 66.85% in cleared plots).

Greatest *P. halepensis* seedling density values were recorded during the first autumn–winter after fire in both the treatments (6.61 ± 2.11 and 8.66 ± 3.97 seedlings/m^2, respectively), and differences were not significant ($t_{38} = 2.00$; $p > 0.05$). Density diminished throughout the study according to mortality. The decline was greatest in cleared plots (Fig. 3). However, four years after the fire, seedling density

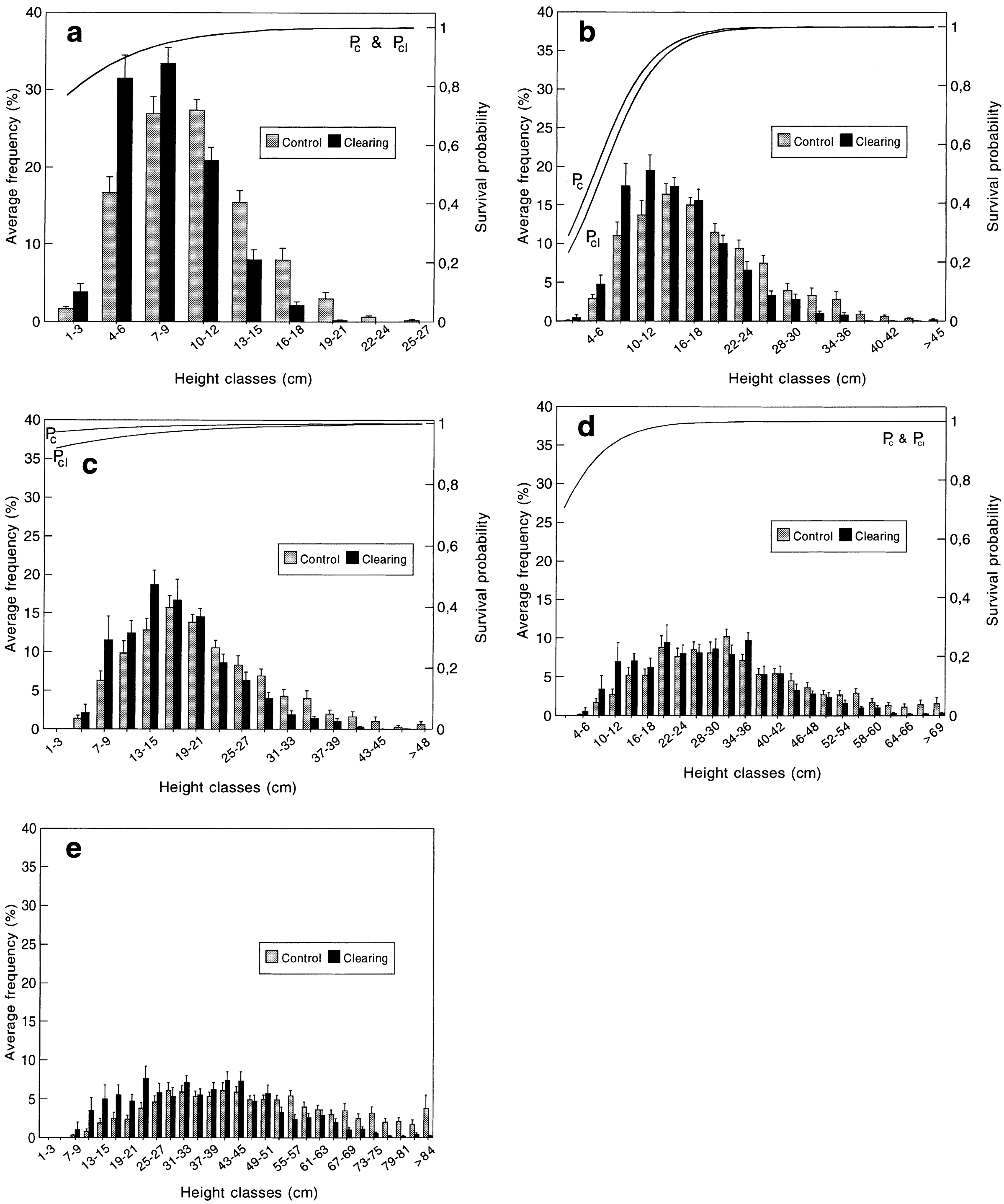

Fig. 2. Average frequency (+SE) of height classes (3 cm intervals) of *P. halepensis* seedlings emerged during the first six months after fire in the control and cleared areas in (a) January 1996, (b) June 1996, (c) January 1997, (d) June 1997, and (e) June 1998. In every graph, seedling survival probability in the next sampling as a function of the current height and treatment is represented by a logistic regression curve. P_c = seedling survival probability in the control area; P_{cl} = seedling survival probability in the cleared area. Equations for significant models are: (a) January 1996: $z = 0.96 + 0.24$ height; (b) June 1996: $z = (-1.19) + (-0.30)$ treatment $+0.29$ height; (c) January 1997: $z = 2.25 + 1.09$ treatment $+0.07$ height; and (d) June 1997: $z = 0.51 + 0.18$ height.

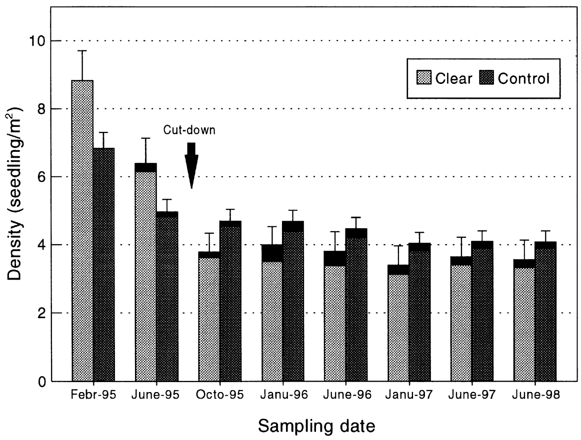

Fig. 3. Changes recorded throughout the study period in average density (individuals/m^2; +SE) of *P. halepensis* seedlings which emerged during the first six months after fire in control and cleared plots. The dark portions in bars denote changes in seedling density caused by individuals which emerged from February 1995.

remained large in both these areas (3.89 $\pm$ 1.48 in control plots, and 3.35 $\pm$ 2.64 in cleared plots), and differences were not statistically significant ($t_{38} = 0.80$; $p > 0.05$). At the end of the study period, 90% to 95% of *P. halepensis* seedlings had been formed by individuals which had emerged during the first six post-fire months (Fig. 3).

4.3. Seedling growth

Average height and seedling growth rates were always significantly greater in control plots than in cleared plots (Table 2). Seedling growth was substantially greater in control than in cleared area between the third and the fourth post-fire years (Table 2). Seedling height distribution showed a more pronounced skew towards the higher height classes in control than in cleared plots (Fig. 2). Over time, the dominance of a few height classes in seedling populations diminished and dispersion of height values increased. This 'height dispersion' was always more marked in the control area. After wood removal treatment, modal height classes were in greater control than in the cleared plots. By contrast, individuals in shorter classes were always more abundant in cleared plots. Four years after the fire, the tallest individual in the control area measured 117 cm, whereas in the cleared area the tallest individual seedling was 99 cm.

5. Discussion

5.1. Seedling emergence

The exact effect of the burnt tree felling and trunk dragging on *P. halepensis* seedling emergence could not be satisfactorily tested in the study, since seed germination of this species was concentrated before the experimental wood removal being carried out. However, the results clearly showed this treatment simulating traditional management practices did not significantly affect *P. halepensis* seedling emergence dynamics. It may be due either (i) to a total seed bank depletion before wood removal or (ii) to the absence of any emergence stimulation effect. Regardless of the reason, the study evidenced that the trunk dragging treatment is not necessary at all for ensuring seed germination, at least in Mediterranean pine forests,

Table 2
P. halepensis seedling height ($\pm$SD) and relative growth (in percentage; $\pm$SD) measured from wood removal to the end of the study. For every sampling, Student *t* test values for height and growth comparisons between control and cleared areas are

Sampling date	Height (cm)			Growth (%)		
	control	clear	t_{38}	control	clear	t_{38}
Jan 1996	10.34 $\pm$ 1.66	8.17 $\pm$ 1.27	4.66^b	—	—	—
Jun 1996	17.89 $\pm$ 4.59	14.59 $\pm$ 2.97	2.69^b	71.99 $\pm$ 19.71	78.21 $\pm$ 16.89	0.66
Jan 1997	20.72 $\pm$ 5.39	17.31 $\pm$ 3.23	2.42^a	13.78 $\pm$ 2.78	11.42 $\pm$ 3.29	2.24^a
Jun 1997	33.02 $\pm$ 9.36	28.33 $\pm$ 6.50	2.04^a	60.02 $\pm$ 7.18	63.61 $\pm$ 1364	1.22
Jun 1998	47.06 $\pm$ 12.51	36.50 $\pm$ 8.52	3.12^b	43.79 $\pm$ 5.16	28.54 $\pm$ 3.48	10.83^c

^{a} $p < 0.05$.
^{b} $p < 0.01$.
^{c} $p < 0.001$.

where the litter layer is completely consumed by fire. Post-fire *P. halepensis* regeneration, both in the control and in the cleared area was in accordance with the general pattern described in the literature for this species (Abbas et al., 1984; Trabaud, 1988; Moravec, 1990; Saracino and Leone, 1993; Thanos et al., 1996; Herranz et al., 1997). *P. halepensis* is an obligate seeder, so that its recovery strategy consists of rapid seedling recruitment immediately after fire. Our results show that seedling emergence was concentrated in the earliest post-fire stage (first six months after fire) which gave rise to large seedling density at the end of this period. However, a second post-fire germination peak was recorded in the second autumn after the fire in both, the cleared and control areas. In spite of its low relative contribution to the whole pine seedling population, this second seed germination period produced an additional average seedling density of ca. 0.2–0.4 seedling/m^2, which was reduced to 0.11 seedling/m^2 in both these areas by the end of the study. Since *P. halepensis* does not accumulate persistent seed reserves in the soil (Daskalakou and Thanos, 1996; Ferrandis, 1996) and no tree survived fire, neither within nor near the permanent plots, the second group of seedlings may have emerged from seeds dispersed from cones opened, but not killed by heat, that is to say, they remained on the trees and shed their seeds during the early years after fire (see Trabaud et al., 1985). In the experimental wood removal area, *P. halepensis* seeds retained in burnt trees may have been released during the cutting down and trunk dragging process.

5.2. Seedling mortality and density

The results show a remarkable effect of burnt wood removal on established seedling survival. Dead trunk dragging operations produced direct seedling mortality (i.e. 34%), and significantly increased in first post-fire summer seedling mortality. Natural seedling mortality was concentrated in first post-fire spring, in contrast to results from most other works on *P. halepensis* regeneration dynamics (Nahal, 1962; Acherar, 1981; Trabaud, 1988; Moravec, 1990; Saracino et al., 1993; Thanos et al., 1996; Herranz et al., 1997) which reported the first post-fire summer as the period with the greatest mortality rate. The reasons for the early mortality in our study may have been the spring drought recorded in that year and previous winter frosts which can debilitate and even kill young seedlings (Trabaud, 1988). The low mortality recorded during the first post-fire summer may be explained by the selection suffered by seedlings during the previous spring. According to Trabaud (1988), seedlings surviving this early critical period achieve a sufficient root development so as to substantially increase survival probability if shrub competition is not intense, as showed by Martínez-Sánchez et al. (1997) for the first post-fire year in the study area. In clear areas, the total absence of canopy cover and the possible debilitation of many seedlings by mechanical action during wood removal operations could increase seedling vulnerability to water stress during the first post-fire summer period. As a consequence, the total cumulative mortality at the end of the study was nearly 74% in the clear area, in contrast to 48% in control plots.

Seedling height was negatively correlated with mortality, as revealed in other studies (Trabaud, 1988; Thanos et al., 1996). In both the areas, seedlings reaching – or exceeding – 15 cm two years after fire had a high survival probability, with mortality occurring mostly in seedlings in shorter height classes. This correlation was significantly increased by wood removal treatment between the second and the third post-fire summers. The greater sunlight exposure could explain the higher mortality of small and summer drought and insolation sensitive seedlings (Nahal, 1962; Gindel, 1964; Acherar, 1981). In the present study, the low frequency of shorter height classes in seedling populations in both these areas after the first post-fire summer implied that no significant difference in total mortality rate was detected in subsequent samplings. However, wood removal may increase fatality where adverse environmental conditions during the early post-fire stages produced seedling populations dominated by critically short classes. In that case, wood removal should be avoided or delayed to allow seedlings to grow and become more resistant.

All dead trees standing in control plots were windthrown from the second post-fire year, many falling on pine seedlings and pulling up large soil-root plates around the trunk base. However, mortality due to this cause was not great (nearly 4%), so that it would not substantially affect post-fire *P. halepensis* regeneration dynamics.

In spite of the negative effect of wood removal on seedling survival, seedling density was large in the cleared area four years after fire (3.35 seedlings/m^2). This value is greater than that found in many burnt areas without post-fire forestry management in the Mediterranean Basin. Such values ranged from 0.1 to 1.5 seedlings/m^2, one or two years after fire (Trabaud et al., 1985; Trabaud, 1988; Moravec, 1990; Saracino and Leone, 1993; Thanos et al., 1996). Thus, the success of post-fire *P. halepensis* re-establishment would be based on the notably high seedling recruitment capacity immediately after fire. Thus, most seedlings recorded at the end of the study (90–95%) had emerged during the first post-fire six months. In mature *P. halepensis* stands with conspicuous seedling recruitment potential, and under suitable environmental conditions during the early post-fire stages, burnt-wood removal has a moderate rather than a drastic effect in quantitative terms on natural *P. halepensis* re-establishment.

5.3. Seedling growth

Significant mortality of short seedlings during the early stages after fire rendered the use of changes in average population height between two consecutive samplings unsuitable as a viable indicator of seedling growth. Selective elimination of shorter seedlings produced an apparent height increase, which was unrealistically large. Individual labelling, allowed us to determine the actual growth of each seedling, and hence, of the population. Seedling height was always significantly higher in the control stand, and differences in seedling height growth rate by wood-removal treatment became particularly noticeable in the fourth post-fire year. At the end of the study, control average seedling height was 11 cm taller than in the treated area. This effect has been interpreted as a consequence of the increase of water stress on seedlings when burnt trees are removed (Saracino and Leone, 1993). It seems clear that dead trees provide a direct sunlight exposure protection for young pine seedlings during the two post-fire years because they remain upright and maintain a nearly intact crown structure which favours seedling development so that the second null hypothesis considered in the study is confirmed.

In spite of this detrimental effect on seedling growth, the results reveal no dramatic effect on *P. halepensis* seedling survival due to burnt pine felling and wood removal if initial seedling density is high enough. Four years after the fire, there were ca. 33 000 saplings/ha in the cleared stand (in mature pine forests in the area with an optimal timber production, tree density ranges from 400 to 600 per ha) with an average height of 36 cm, and seedling mortality became <3%. It seems reasonable to assume that, under such early post-fire stage conditions, the natural regeneration of the pine forest originally installed in the area is ensured. Future studies focused on other ecological aspects should be investigated in order to gather the necessary information for a correct burnt-pine forest management. The effect of a reduction in seedling growth rate on the final tree development level, the effects of dead wood non-removal as a possible source of proliferation of xylophagous arthropod outbreaks and as an accumulation of plant fuel which increases wildfire risks, and the implications for the biogeochemical cycles, need investigation in detail. The increase in the ecological-niche diversity and the protection offered to animal communities by the accumulation of woody debris (two years after fire all the trees were windthrown) in uncleared burnt areas should also be contrasted with the negative effects of burnt-wood removal pointed out by Llimona et al. (1993) and Izaki (1993).

6. Conclusions

1. Burnt-tree removal is not necessary to ensure seedling emergence and post-fire re-establishment of *P. halepensis*. High seedling recruitment during the earliest stages (six months) after fire guarantees natural regeneration of the pine forest.
2. Wood removal increases seedling mortality, but does not threaten *P. halepensis* natural re-establishment when the initial seedling density is high.
3. Wood removal produces a moderate reduction in seedling growth; its effect on the final tree-development level needs investigations.

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Appendix

Nomenclature

For nomenclature see Tutin et al. (1964).

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