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Fire severity and seedling establishment in *Pinus halepensis* woodlands, eastern Iberian Peninsula

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Abstract

Given the observed heterogeneity in fire severity produced within wildfires, we asked to what extent this heterogeneity might affect post-fire regeneration. For this purpose, we studied the post-fire dynamics of *Pinus halepensis* (Aleppo pine) in the eastern Iberian Peninsula. Sampling was stratified on the basis of fire severity. We defined three fire severity classes based on the degree of consumption of the pine canopy. The results suggested that there is no clear relationship between seedling density and fire severity; however, mortality was lower and growth (height, shoot biomass and root biomass) was higher in the high severity class. These results can be explained by soil processes: Sites in the high fire severity class may have sustained higher fire intensities, resulting in higher soil organic matter mineralisation and higher ash deposition, and thus in higher post-fire soil fertility. This higher fertility would produce faster growth in pine seedlings. Independent of the severity class, seedling mortality was higher in quadrats (50 × 50 cm) with higher cover of the perennial grass *Brachypodium retusum* (Poaceae), suggesting a possible competitive effect. For all plots in all 3 severity classes, spatial analysis suggests an aggregate seedlings pattern, but with independence from the position of the adult (source) trees.

Introduction

The regeneration of *Pinus halepensis* woodlands after fire has been studied elsewhere (e.g., Trabaud et al. (1985a, 1985b) and Moravec (1990), Thanos et al. (1996), Daskalakou and Thanos (1997), Ne'eman (1997), Herranz et al. (1997), Tsitsoni (1997), Pausas et al. (1999), Arianoutsou and Ne'eman (2000), Trabaud (2000), Leone et al. (2000)). However, due to topography (Takaoka and Sasa 1996), wind and microclimatic changes during a fire (Gómez-Tejedor et al. 1999), and fuel heterogeneity (Miller and Urban 2000), both the intensity and the severity of fires are quite variable throughout the landscape and produce heterogeneous post-fire environments. Thus, fires contain areas with different fire intensities and severities, usually in a complex mosaic pattern (van Wagner 1983; Turner et al. 1994). Despite the importance of fire in the Mediterranean basin, the different post-fire regeneration patterns (regeneration variability) in patches with different degrees of consumption (fire severity) have not yet been addressed. In this framework, the present study aims to examine the effect of fire severity on the regeneration of *Pinus halepensis.*

P. halepensis is a dominant tree in the western Mediterranean basin, and especially in the eastern Iberian Peninsula (Trabaud 2000). In this area, the number of wildland fires is growing despite increasing fire-fighting resources (Piñol et al. 1998; Pausas and Vallejo 1999; Pausas 2002). However, the annual surface burnt fluctuates greatly and is more related to the annual climatic conditions than to the number of fires (Pausas 2002); that is, the number of fires increases every year but large fires occur only in dry years. In this framework of high fire frequency, information about fire-response variability in pine woodlands is essential for prediction of post-fire regeneration and efficient management.

There is some confusion in the literature over the concepts of fire intensity and fire severity, so some clarification is given here (Brown and DeByle 1987; Hartfort and Frandsen 1992): Fire intensity is the rate of energy or heat release per unit time per unit length of fire front (Byram 1959), kW/m; it is related to flame length. Fire severity is the effect of fire on an ecosystem, that is, on living plants, as well as on the amount and location of organic matter consumed during a fire (Ryan and Noste 1985). Fire intensity and fire severity may or may not be related.

Unlike fire intensity, there is no widely accepted or standardised quantitative measure of fire severity; quantitative descriptions based on the degree of fuel consumption are common (e.g., Morgan and Neuenschwander (1988) and Bradley et al. (1992), Turner et al. (1994)). Other researchers have used the degree of vegetation mortality as a measure of fire severity (Atkin and Hobbs 1995; Chappell and Agee 1996).

In the present work, we classify fire severity according to the degree of consumption of the canopy after a crown fire. Using this approach, Turner et al. (1994) provided a classification of fire severity in three classes: light surface burn (unaffected canopy), severe surface burn (scorched canopy leaves) and crown fire (consumed canopy leaves). In most of our burnt areas, pine woodlands fall into the latter two categories (i.e. light surface burns are very rare), and depending on the scorched height, trees may or may not survive. Thus, we decided to divide the second type from Turner et al. (1994) into two classes (see methods for details). This measure of fire severity, which is related to flame height, provides information on fire severity at the canopy level. To what extent fire severity at the canopy is related to fire severity at ground level remains unknown; however, the fact that may be necessary a certain amount of fuel to produce long flames suggest that fire severity and intensity could well be related, although the effect of topography and wind may disrupt this relationship. However, we can hypothesise that post-fire *P. halepensis* regeneration may be related to fire severity (as defined above) because: a) *P. halepensis* has serotinous cones and fire at canopy level may control seed release (Saracino 1997) and mortality; b) different canopy consumptions may create different post-fire litter inputs from the scorched canopy; and c) different fire severities may produce different accumulations of ashes and changes in soil nutrient availability.

Methods

Study area

The study was carried out on Ferradura Mountain (200–500 m a.s.l.; UTM 31TYK54), in the municipality of Cabanes (Plana Alta, Castelló, Spain), in the eastern Iberian Peninsula. The climate is typically thermo-mediterranean with a mean annual temperature of about 16 °C and an average annual rainfall of 486 mm (Cabanes Meteorological Station, 291 m a.s.l.; 1989–1990) and 540 mm (Vilafamés Meteorological Station, 295 m a.s.l.; 1973–1990). The predominant bedrock type on the slopes is carbonated marls from the Cretaceous, while the upper part of the mountain is composed of hard limestone. As in many other areas in Spain (Pausas and Vallejo 1999; Pausas 2002), most of the slopes were terraced during ancient times and abandoned later during industrialisation. Current vegetation on these slopes consists of *Pinus halepensis* (Aleppo pine) woodlands with an understorey composed of numerous shrubs (*Erica multiflora*, *Pistacia lentiscus*, *Rhamnus alaternus*, *R. lycioides*, *Lonicera implexa, Anthyllis cytisoides, Rubia peregrina, Asparagus acutifolius, Smilax aspera, Daphne gnidium, Globularia alypum, Ulex parviflorus, Chamaerops humilis, Juniperus oxycedrus, Rosmarinus officinalis*). The herb layer is dominated by the perennial grass *Brachypodium retusum*. At the upper part of the mountain (on limestone), the dominant vegetation is *Quercus coccifera* garrigues.

The whole area (ca. 900 ha) was burnt by a crown wildfire in April 1999, leaving a mosaic of pine woodland patches with different degrees of consumption. The area had previously been affected by another wildfire in 1979, from which many trees had survived (low severity fire). Thus, before the 1999 fire, the pine population consisted of a ca. 20-year-old regeneration population plus a previous population of older trees.

Sampling

After the 1999 fire, three different fire severity classes were differentiated according to pine canopy damage: low, moderate and high (see Table 1 for detailed definition).

Four permanent plots for each fire severity class (i.e. 12 plots) were selected on Ferradura Mountain. Each of these rectangular plots was situated on a flat area of an old terrace. Plot size ranged from 81.5 to 145 m^2 (mean = 109 m^2) depending on the site con-

Table 1. Classes of fire severity judged from *Pinus halepensis* canopy damage; description and post-fire mortality of the adult pines. In all three classes, organic litter layer was consumed.

Fire severity classes	Description	Post-fire mortality
Low	Light fire; canopy trees retain > 20% of green leaves (top of the canopy). Trees remain mainly green after the fire.	No
Moderate	Most leaves (> 80%) of canopy trees are scorched (dead) but not consumed. Green leaves may occur on the top (< 5%), and some leaves at the bottom may be consumed (< 5%). Trees are mainly brown (retained scorched leaves) after the fire.	Yes
High	Severe fire; canopy trees with > 80% of the leaves consumed and the rest (if any) scorched (top). No green leaves left.	Yes

strictions (continuity of canopy, size and shape of the terrace, homogeneity of conditions). Diameter at breast height was measured in all the stand trees of each plot and in all the adjacent trees whose projected canopy area fell within the plot. The proportion of projected canopy area that fell within the plot was also recorded. Vegetation and litter cover were sampled in 12/1999 using 20 50 × 50 cm quadrats in each plot. Of these 20 quadrats, 10 were randomly distributed in the plot, and 10 were centred on 10 randomly chosen pine seedlings. Seedling emergence and survival were sampled in 12/1999, 5/2000, 11/2000, and 9/2001, that is, 8, 13, 19 and 29 months after the fire. All seedlings were referenced within a 2-dimensional Cartesian system to permit spatial analysis. During the last sampling (29 months after fire), seedling height and basal diameter were also measured, and 10 seedlings of high severity and 10 of low severity were collected (above- and below-ground). The number of burnt resprouting shrubs that failed to resprout 13 months after fire (i.e., one year post-fire) was also quantified for each plot.

Analysis

On each plot, tree density and basal area of the adult trees were computed. In order to minimise the edge effect, the basal area was computed to include the individual trees adjacent to the plot weighted by the proportion of the canopy area covering the plot (weighted basal area). Seedling density and mortality, vegetation cover and litter cover were calculated for each plot. Collected seedlings were separated in different fractions (roots, aerial woody parts, leaves), oven-dried (60 °C for 48 h) and weighed. All this information was evaluated by averaging the data from plots of the same fire severity class and performing analysis of variance (ANOVA). When ANOVA was significant, pairwise multiple comparison was also applied (Fisher's LSD). Previous to the statistical analysis, the data were tested for Normality and for homogeneity of variances, and the following transformations were applied: logarithmic for biomass and height values; arcsinus of squared root for proportion.

For each plot, the pine seedling spatial pattern was analysed by the Ripley's *K*-function (specifically, $\sqrt{[K(t)/\pi]}-t$; Ripley (1979, 1981) and Haase (1995)). The univariate (single-species) test was used to test the randomness of the seedling spatial distribution, and the bivariate test was used to test the randomness of the spatial relationship between seedlings and adults.

Results

The three sets of plots related to the three fire severity classes showed different adult pine density; however, the mean diameters and the weighted basal area (potential seed source) were similar ($p > 0.10$) for the three fire severity classes (Table 2), ensuring that the plots were comparable.

Eight months after the fire, the amount of litter on the soil was very low in the high fire severity plots (< 5%), and much higher (ca. 45%) in the moderate and low severity plots (Figure 1). At the same sampling

Table 2. Some characteristics of the canopy trees and saplings of cohort I for the plots in low, moderate and high fire severity (*, $p < 0.05$; ***, $p < 0.001$; ****, $p < 0.0001$; ns, not significant).

	Fire severity class			ANOVA
	Low	Moderate	High	
Pre-fire canopy trees				
– Mean diameter (cm)	10.24	9.59	10.09	ns
– Standard deviation of the diameter (cm)	7.48	6.65	9.57	–
– Weighted basal area (m^2/ha)	21.6	23.5	21.8	ns
Sapling of cohort I (29 months after fire)				
– Height (cm)	14.3	17.2	23.5	****
– Diameter (cm)	0.21	0.24	0.42	****
– Number of branches (#)	4.0	3.7	10.7	***
Resprouting failure (%)	30.6	39.9	61.2	*

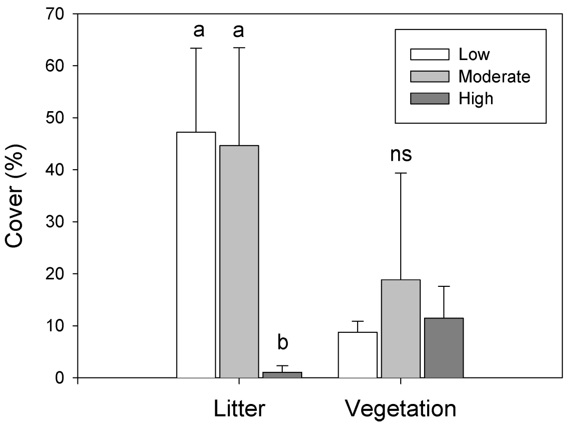

Figure 1. Percentages of the soil covered by litter and by vegetation in the three fire severity classes 8 months after the fire. Columns with different letters are significantly different ($p < 0.05$).

time, vegetation cover (total cover, herbaceous cover) did not show any relation to the fire severity class.

The first post-fire seed germination was observed in autumn 1999, which corresponds to the period with the first important rainfalls after the fire. From the total of 489 individuals tagged on the first sampling date (cohort I) throughout the whole study site, 182 (37%) survived after 2.5 years (Figure 2a). The survival for cohorts II and III was 48 and 92% respectively. For cohort I (the largest in number), the mortality rate was approximately linear with time during the first year after emergence (19 months post-fire), and then mortality remained very low. This pattern was observed for all the 3 fire severity classes (not shown). Mortality tended to be lower for the high severity than for the low and moderate fire severity classes; however, the large variation in mortality between plots and the low number of plots makes the differences not significant (Figure 2b).

Considering both emergence and mortality, seedling density ranged from 0.1 to 1.43 (mean = 0.45) individuals/m^2 in the first sampling (8 months after fire) and from 0.01 to 0.52 (mean = 0.27) 2.5 years after fire. The highest seedling density was found in the low severity class and the lowest in the moderate (intermediate) severity class, although a large variation was found (Figure 3). The high fire severity plots showed the most stable seedling density with time (Figure 3), due to the lower mortality indicated above (Figure 2b).

Two-and-a-half years after the fire, seedlings in the high fire severity class were significantly taller (Figure 4a), had thicker stems and a higher number of branches (Table 2) than those in the low and moderate fire severity classes. Consequently, biomass values (both above- and below-ground) were significantly ($p < 0.01$) higher for the high severity class than for the low one (Figure 4b), although the root:shoot ratio was similar.

Cover of the perennial grass *Brachypodium retusum* was significantly higher in the quadrats (50 × 50) in which the seedling died than in those where the seedling survived (Figure 5), suggesting that seedling mortality may be related to grass cover.

The estimation of Ripley's *K*-functions suggested that, for all 12 plots, the spatial relationship between trees and seedlings was random (Figure 6a), that is, seed germination was not related to the location of seed-source trees. However, analysis of the spatial pattern of seedlings only (single-species test) showed that, in all plots, seedlings germinated in an aggregate pattern (Figure 6b). Neither of the two *K*-functions types studied showed a clear relationship between spatial pattern and fire severity class.

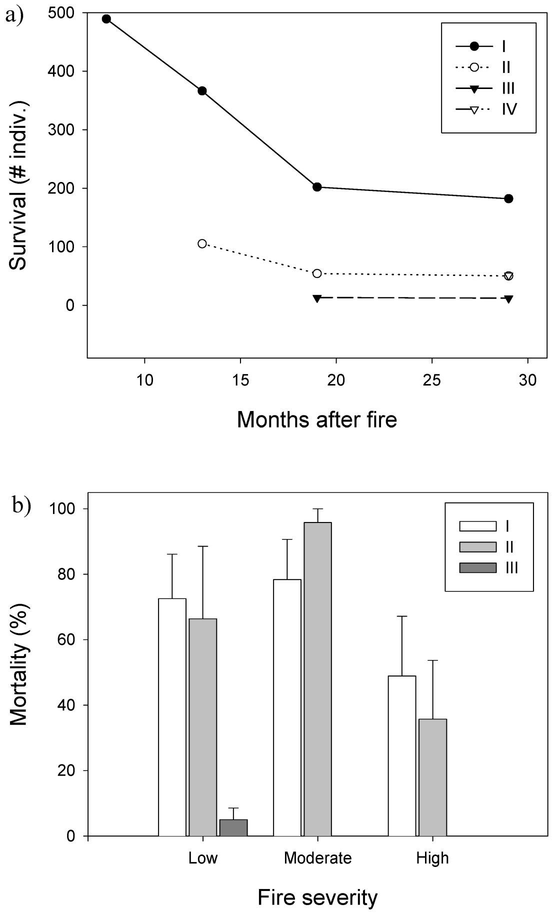

Figure 2. a) Total number of pine seedlings surviving through time (different symbols are the new cohorts for each sampling date). Note that the number of new individuals in the last sampling (cohort IV, 29 months after fire) is about the same as the number of surviving individuals from cohort II. b) Seedling mortality (% of individuals) of each cohort in the different fire severity classes. For any cohort, ANOVA did not show significant differences (cohort I: F = 1.18, p = 0.34; cohort II: F = 2.49, p = 0.15; cohort III: F = 0.60, p = 0.65).

Resprout failure in woody species increased with fire severity (Table 2). Woody resprouters were mainly: *Erica multiflora, Pistacia lentiscus, Rhamnus alaternus, R. lycioides, Anthyllis cytisoides, Daphne gnidium, Globularia alypum, Chamaerops humilis* and *Juniperus oxycedrus*.

Discussion

Our results suggest that different fire severity classes, defined by the canopy damage, produce different post-fire soil conditions; that is, there was much less

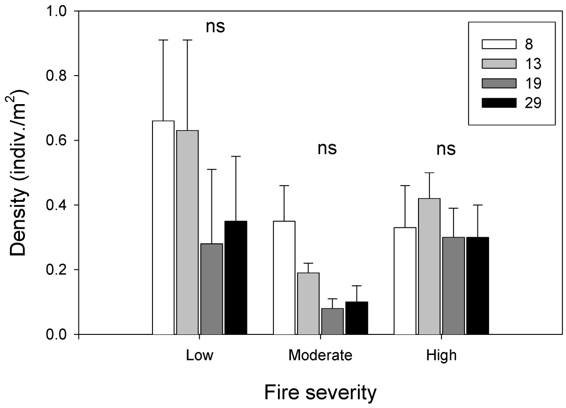

Figure 3. Changes in average seedling density (individuals/m^2) 8, 13, 19 and 29 months after fire for the three different fire severity classes (ANOVA did no show any significant difference: F = 1.48, p = 0.27; F = 3.21, p = 0.07; F = 0.27, p = 0.84; for low, moderate and high fire severity classes, respectively).

litter in the high severity fire than in the moderate and low severity ones. This indicates that different fire severities produce different soil conditions (e.g., different mulching effects and organic matter inputs; Morgan (1986) and Mando et al. (1999)) which may have implications on the emergence of seedlings. Initial pine seedling densities tended to be higher in the low severity fire; however, seedling mortality was lower and growth was higher in the high fire severity class. Thus, the final (2.5 years) seedling densities were not related to fire severity. In most plots, seedling density values were low compared with those found in most sites from Greece and the Near East (Tsitsoni 1997; Ne'eman et al. 1992; Eshel et al. 2000), but they were similar to those found in south-eastern Spain (Herranz et al. 1997).

The fact that the overall mortality rate remained strong and almost constant during the first year after emergence may be due to interspecific competition, especially with *Brachypodium retusum* (the dominant perennial grass in the study area), and also to the typical summer drought and late-summer torrential rainfalls in the eastern Iberian Peninsula (Pérez Cuevas 1994; Peñarrocha et al. 2002). The possible effect of grazing was not studied but cannot be denied.

The aggregated spatial pattern of seedlings has frequently been discussed in the literature for many different ecosystems (Hubbell 1979; Okuda et al. 1997; Debski et al. 2000). We did not find any association between seedlings and mother trees, but we did find a clear pattern of seedling aggregation. The study by Ne'eman and collaborators in *Pinus halepensis* wood-

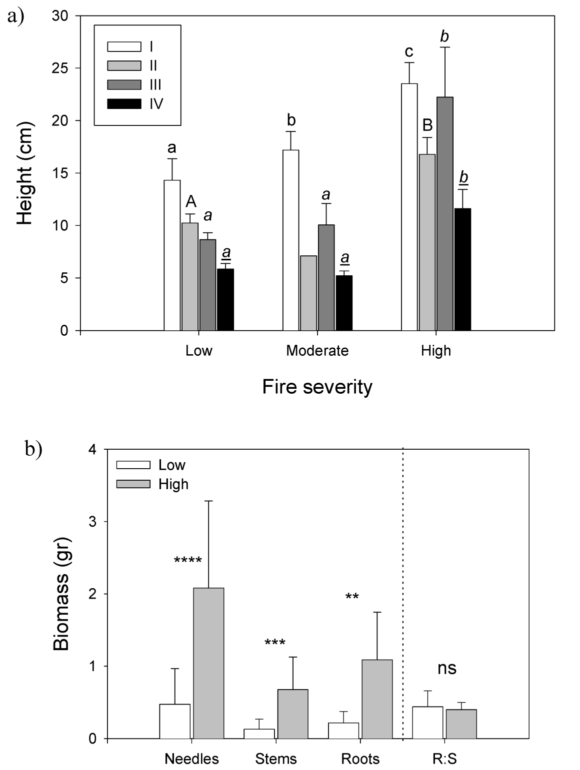

Figure 4. (a) Mean seedling height (cm) for each cohort and for each severity class at 29 months after fire. For cohort III in high severity and II and III in moderate severity, the number of seedlings was very low (< 3). For each cohort, columns with different letters are significantly different ($p < 0.05$). ANOVA results for each cohort: cohort I: $F = 29.59$, $p < 0.0001$; cohort II: $F = 6.09$, $p = 0.004$; cohort III: $F = 10.25$, $p = 0.003$; cohort IV: $F = 8.24$, $p = 0.001$. (b) Mean biomass (gr) of different fractions of pine saplings (leaves, aboveground woody parts and roots), and root/shoot biomass ratio (R/S), for saplings of cohort I growing in the low and the high fire severity classes 29 months after fire. Differences between the low and the high severity classes are significant for all fractions (****, $p < 0.0001$; ***, $p < 0.001$; **, $p < 0.01$) but not for the R/S (ns).

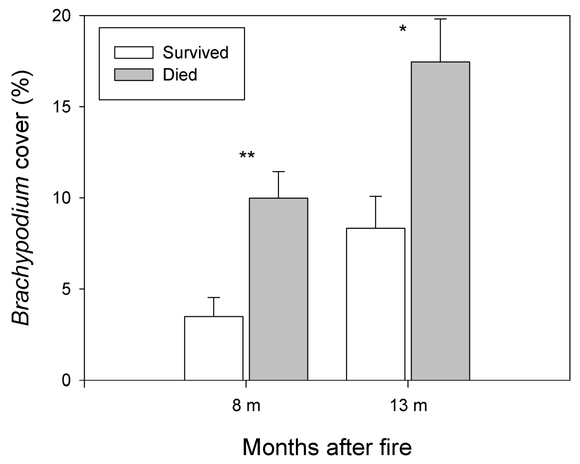

Figure 5. Cover of the perennial grass *Brachypodium retusum* in the 50 × 50 quadrats where the pine seedlings (of cohort I) survived and in those where the seedlings died (ANOVA for 8 months after fire: $F = 8.32$, $p = 0.005$; and for 13 months after fire: $F = 4.25$, $p = 0.042$).

lands of the Near East (Ne'eman et al. 1992; Ne'eman 1997) suggested that the spatial relation of seedlings and saplings with adults may change with time. They found that during the first period (1–4 years after fire) the aggregation was low, but later (5–20 years after fire) there was an increased aggregation pattern due to the differential mortality in relation to the distance to the burned tree (higher survival and growth close to the burned tree).

Seedling growth was clearly higher on the high severity plots, but the root:shoot ratio was equal. This indicates that there was a constant resource allocation to roots and shoots even when the growth and biomass values were very different.

The question that arises from the results is why pine seedlings at high fire severity had higher growth than those submitted to low fire severity, given that at low fire severity there was more litter in the soil (i.e., mulching effect and input of organic matter). The answer may lie in the changes in the mineral soil caused by the different fire severities. In the *Pinus halepensis* woodlands of the Near East (Ne'eman et al. 1992; Ne'eman 1997), seedling mortality was lower and seedling growth faster near the burned tree with higher fire intensity and ash deposition; a result that fully agrees with our results.

Although soil data on post-fire nutrient availability were not available for our sites, foliar analysis of the needles that grew during the first post-fire year (cohort I) may provide some clues on the possible soil nutritional status of the soil at that time (Binkley 1986). There was a tendency toward higher phosphorus concentrations in the higher fire severity class (0.519 ± 0.17 and 0.690 ± 0.22 mg/g for low and high fire severity classes; Pausas et al. in press), but this is not the case for nitrogen concentrations (1.07 ± 0.16 and 1.08 ± 0.11% for low and high fire severity classes; Pausas et al. (2002)). Serrasolsas and Vallejo (1999) studied post-fire soil nutrient dynamics in a *Quercus ilex* community of the eastern Iberian Peninsula and showed an increase in both soil available phosphorus and nitrogen during the first post-fire

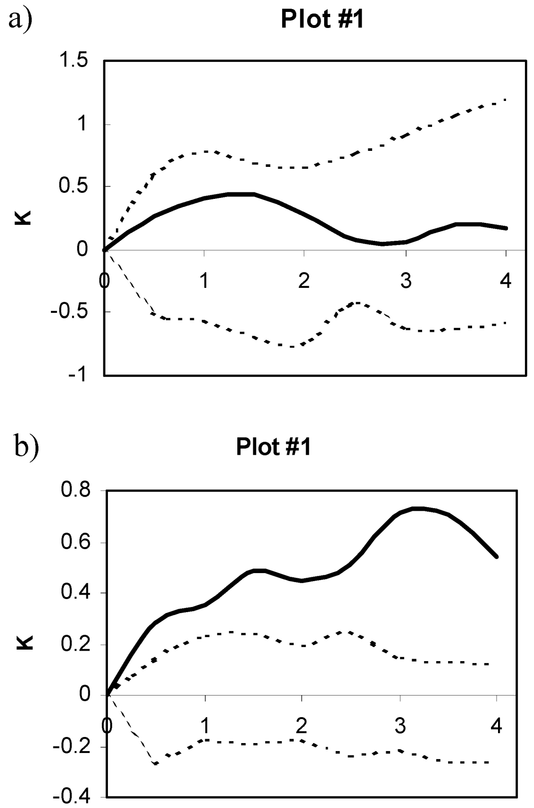

Figure 6. Examples of Ripley's K-functions (thick line) for one of the plots (number 1). Dashed lines indicate the limits of significance. a) Bivariate (trees – seedling relationship), b) Univariate (seedling spatial pattern). Line within the significant limits indicates randomness (a); line above the significant limits indicates aggregation (b).

year. Romanyà et al. (2001) also showed higher soil nutrients in plots with higher fire severity (higher soil heating by an experimental fire) in Mediterranean old fields, although in their case they only studied soil mineral N. However, other studies found a decrease in N mineralisation in burned vs. control plots (Ojima et al. 1994). The increase or decrease in N availability after fire was not easily related to fire severity because the higher the severity the higher the N losses by volatilisation, but also the N mineralisation by heating the top soil. On the contrary, the amount of P lost by volatilisation is usually low and the amount P available to plants is more related to ash deposition than N (Raison et al. 1985). For the case in which mineral N was higher in our high severity plots, foliar analysis indicated that the seedlings were not nitrogen deficient. In fact, the needle N concentration found in our seedlings (ca. 1.1%) is not very different from the N concentration observed in pine seedlings growing in nursery conditions (1.1–1.7%, Valdecantos (2001)), while our values of total P (ca. 0.5–0.7 mg/g) are much lower than those found in nursery seedlings (3.1–3.5 mg/g, Valdecantos (2001)). Furthermore, in our area, forest soils on marls are typically more deficient in P than in N (Valdecantos 2001).

The study by Serrasolsas and Vallejo (1999) also showed that in a clear-cut (unburned) plot where the litter and branches were left on the forest floor, the increase in nutrients was much lower than in a burned plot. Their burned plot can be compared with our high severity plot in the sense that all the leaves were consumed by the fire, while their clear-cut plot may be compared with our low severity plots in the sense that organic matter was left on the soil after removing all standing biomass. Within this framework, we suggest that the higher nutrient availability in the high fire severity class may explain the higher growth of the seedlings.

The fact that plots with different canopy damage seem to show different soil nutrient dynamics and different resprouting failure suggests that fire severity at the canopy level may be related to fire intensity at the ground level. Different canopy damage is produced by different flame heights, which are related to the understorey fuel load. Thus, high fuel loads may produce high fire intensity to the soil and stumps and also tall flames. This also suggests that post-fire regeneration may be related to pre-fire landscape variability (e.g., variability of fuel loads).

In conclusion, in our study area, post-fire regeneration of *Pinus halepensis* seems to be controlled by the changes in soil fertility (especially P) via ash accumulation and increased mineralisation. In the short term (ca. 2.5 years after fire), better regeneration of pines is found at high fire severity because of the increased post-fire nutrient concentration. The implication for long-term vegetation dynamics could be related to the change in fitness between seeders and resprouters (Bellingham and Sparrow 2000; Pausas 2001) due to different fire severity. Our results suggest that high fire severity favours seeder species (because of the increased soil nutrient availability and consequent seedling growth, Figure 4) versus resprouters (because of higher mortality with fire severity, Table 2); while low fire severity would favour resprouter species (lower mortality). This is in agreement with the lower resprouting allocation at high fire severity suggested by Bellingham and Sparrow

(2000). However, further investigations are needed in this context before we can generalise this prediction.

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