

Growth and structure of a young Aleppo pine planted forest after thinning for diversification and wildfire prevention

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Abstract

Aim of study: In the Mediterranean, low timber-production forests are frequently thinned to promote biodiversity and reduce wildfire risk, but few studies in the region have addressed such goals. The aim of this research was to compare six thinning regimes applied to create a fuelbreak in a young Aleppo pine (*Pinus halepensis* Mill.) planted forest.

Area of study: A semiarid continental high plateau in south-eastern Spain.

Material and methods: Three thinning intensities (Light, Medium and Heavy) were combined with two thinning methods: i) Random (tree selection), and ii) Regular (tree spacing). Tree growth and stand structure measurements were made four years following treatments.

Main results: Heavy Random thinning successfully transformed the regular tree plantation pattern into a close-to-random spatial tree distribution. Heavy Regular thinning (followed by the Medium Regular and Heavy Random regimes) significantly reduced growth in stand basal area and biomass. Individual tree growth, in contrast, was greater in Heavy and Medium thinnings than in Light ones, which were similar to the Control.

Research highlights: Heavy Random thinning seemed the most appropriate in a young Aleppo pine planted forest to reduce fire risk and artificial tree distribution simultaneously. Light Regular thinning avoids understocking the stand and may be the most suitable treatment for creating a fuelbreak when the undergrowth poses a high fire risk.

Key words: diversity; forest structure; Mediterranean; *Pinus halepensis*; tree growth; wildfire risk.

Introduction

Thinning is a mid-rotation forest management technique that has been widely studied and is commonly applied to optimise timber production and economic return. Its effect on forest structure and diversity has also been the focus of research (Lei *et al.*, 2007; Crecente-Campo *et al.*, 2009). In fact, forest structural diversity is often considered an indicator of biodiversity (Lähde *et al.*, 1999; Pommerening, 2002), due to the known link between forest structure and the habitats provided for other plant, animal or fungi species. Attempts to characterise forest structure have led to the development of various different indices describing the spatial arrangement of forest elements (mainly trees), the mixture and diversity of species, or the degree of differentiation among individuals (Pommerening, 2002; Del Río *et al.*, 2003). In the context of climate change, thinning is being revisited to assess its influence on carbon fixation (Del Río *et al.*, 2008a) and

on improving forest adaptation to future climate scenarios (Cotillas *et al.*, 2009).

However, the possibilities of managing forests through thinning are economically limited in areas with low timber production, such as the Mediterranean Basin. Indeed, Mediterranean forests in southern Europe are often unprofitable nowadays (in terms of marketable products), and this has led to lack of management, a greater degree of abandonment and exposure to risks such as wildfire (Scarascia-Mugnozza *et al.*, 2000; Valbuena-Carabaña *et al.*, 2010). Many of these forests, particularly coniferous high forests, stem from fairly recent degraded-land restoration programmes (Valbuena-Carabaña *et al.*, 2010) and nowadays their management relies largely on scarce public funding, which is focused on the protection of forests from wildfires and the promotion of their social and ecological values (*e.g.*, recreation, biodiversity, carbon storage) (Osem *et al.*, 2008).

One of the most commonly applied wildfire prevention measures is the creation of shaded fuelbreaks (Agee *et al.*, 2000) by thinning and pruning the forest in strategic areas to reduce the amount and continuity

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of vegetation (Graham *et al.*, 1999; Vélez, 2009). Such actions constitute good opportunities to accomplish other management goals in these Mediterranean planted coniferous forests (MPCF, Osem *et al.*, 2008). For example, thinning can be employed to reduce artificiality in MPCF and promote the development of near-natural forests (Ginsberg, 2006; Crecente-Campo *et al.*, 2009), while modifying stand structure locally can increase the diversity at the larger forest scale (Lähde *et al.*, 1999). Thinning should also encourage the growth of larger and stronger trees, as competition for water and nutrients is reduced (López-Serrano *et al.*, 2005). These larger trees are more likely to withstand future surface wildfires (Fernandes *et al.*, 2008) and store a larger canopy seed bank (González-Ochoa *et al.*, 2004; Verkaik and Espelta, 2006), two assets which would improve, respectively, the resistance and resilience of the stand to wildfires.

Unfortunately, there has been little research into thinning of certain important Mediterranean tree species, such as the Aleppo pine (*Pinus halepensis* Miller) (Montero *et al.*, 2001). This is a low demanding and easily regenerated species which has been widely used for afforestation in the Mediterranean Basin, particularly in semi-arid and degraded areas. Even-aged stands of Aleppo pine resulting from plantation or wildfires differ from natural uneven-aged stands. The latter have a complex structure characterised by low tree density (Osem *et al.*, 2008) and numerous clearings (Montero *et al.*, 2001). They may also have a multi-strata structure and include a mixture of other species (Ciancio *et al.*, 2007; Osem *et al.*, 2008).

Generally, early heavy thinning is recommended for Aleppo pine forests, so trees can develop with less competition and reach a larger size (Ne'eman *et al.*, 1995; Montero *et al.*, 2001). However, published research on thinning in these forests has mostly dealt with post-fire regenerated stands (Ne'eman *et al.*, 1995; González-Ochoa *et al.*, 2004; Verkaik and Espelta, 2006) or appears as part of production tables and yield models (Montero *et al.*, 2001). Overall, there is a lack of data from thinning experiments on Aleppo pine MPCF in the literature and more research is needed to identify the best approaches to their management. To this end, we have studied tree growth and stand structure after thinning in a young planted Aleppo pine forest in south-eastern Spain.

The thinning experiments were conducted to create a fuelbreak in association with a wildfire prevention pilot project. Three thinning intensities were combined

with two systematic thinning methods: i) random (tree selection), and ii) regular (tree spacing). We hypothesised that random thinnings would reduce the artificiality in the spatial pattern of trees, resulting in a more natural tree distribution. On the other hand, we expected regular thinnings to maximise tree growth, as even spacing among neighbouring trees would minimise competition for resources. Heavier thinning intensities were hypothesised to increase the effects of the thinning methods.

The objective of this study was to compare these thinning regimes in terms of: i) tree size and growth in the four years after thinning, using standard parameters and locally developed regression equations to estimate tree biomass; and ii) stand structure, using various indices to characterise tree spatial distribution and size differentiation. The analysis of the results is augmented by discussing the likely implications of thinning regimes for wildfire prevention, a major concern in MPCF.

Material and methods

Study site

The study was conducted on the “Cortijo Conejo y Albarrán” estate (Guadix, Granada), in south-eastern Spain (37° 23' N and 3° 03' W, at 1,100 m a.s.l.). Mean annual precipitation is 302 mm, ranging from 204 mm to 370 mm in the study period (2005-2010). In winter the temperature may drop to as low as -15°C and in summer it may reach 40°C. According to Rivas-Martínez and Loidi (1999), the area is in the semi-arid meso-Mediterranean bioclimatic belt. The estate extends across near-flat (2-3% slopes) terrain. Under these environmental conditions, holm oak [*Quercus ilex* L. subsp. *ballota* (Desf.) Samp.] forest could potentially grow on this land. The Aleppo pine is naturally present in the surroundings, particularly on steeper slopes and where there are poorer soil conditions.

This estate was cultivated for decades before it was partially afforested with Aleppo pines in 1994-1996. Trees were planted to a density of 1,500-2,000 trees ha⁻¹, in evenly spaced rows. The forest is traversed by a 35-m-wide and 1.4-km-long firebreak, which is devoid of woody vegetation. In 2005, when the pines were roughly 10 years old, a 38-ha shaded fuelbreak was created to reinforce the existing firebreak. This was accomplished by thinning the plantation alongside

the firebreak, an operation that ended in June 2005. From December 2007 to the end of the study, pine trees were severely affected by a seasonal winter infestation by the pine processionary moth (*Thaumetopoea pityocampa* Den. & Schiff.).

Regression equations

During the thinning, tree measurements were made of 32 Aleppo pines selected randomly from the whole range of tree sizes available in the area ($1.2 \text{ cm} < \text{Diameter at breast height (DBH)} < 10.9 \text{ cm}$; $2.1 \text{ m} < \text{Total height} < 5.4 \text{ m}$). The DBH was calculated as the geometric mean of two-crossed trunk diameters measured at 1.30 m using forest callipers (1 mm precision); while total height was measured by means of a telescopic measuring rod (5 cm precision). These trees were felled and their biomass was divided into three fractions: 1) the trunk, to a diameter (d) of 4 cm at the top end; 2) the branches, with d between 1 cm and 4 cm; and 3) the fine fraction, comprising needles and small branches with d less than 1 cm. Large portable field scales (50 g precision) were used to measure the fresh weight of all fractions. In addition, six samples from each fraction were weighed using a field dynamometer (5 g precision), fully oven-dried (60°C), and weighed again using higher precision laboratory scales. The six ratios between dry and fresh weight calculated for each fraction were averaged and used as conversion coefficients. Moisture content of each fraction was assumed to be similar in all trees.

This 32-tree dataset was employed to build regression equations between the trunk, branches, fine fraction and total above-ground dry biomass (dependent variables), and tree DBH and total height (independent variables). The additivity (Parresol, 1999) of the tree-fraction regression equations was guaranteed by using the statistical procedure described by Ruiz-Peinado *et al.* (2011), which is summarised below.

In a first step, 16 different linear and non-linear equations were calculated for each biomass fraction, and the best model was selected based on graphical analysis of residuals and fitting statistics. For similar accuracies, regressions with a single predictor variable were preferred to multiple regressions. In a second step, the best models selected for each of the fractions were simultaneously fitted using joint generalised regression to make consistent estimates of the different components, so as to meet the additivity property.

Weighted regression was used to correct for heteroscedasticity, frequently present in biomass data. Indeed, observations were weighted using a factor which was estimated through a power function of an independent variable. For further methodological details, please see Ruiz-Peinado *et al.* (2011).

Experimental design

In the creation of the fuelbreak, two experimental factors (thinning method and intensity) were combined to produce six different thinning regimes, which were later compared with each other and with a non-thinned Control area. Two thinning methods were employed: 1) Random (RND), where trees to be felled were randomly selected; and 2) Regular (REG), where equal spacing between residual trees was sought. In addition, three different thinning intensities were tested: 1) Light (L), preserving about 800 trees ha^{-1} ; 2) Medium (M), preserving about 400 trees ha^{-1} ; and 3) Heavy (H), preserving about 250 trees ha^{-1} .

Combining thinning method and intensity required the calculation of three mean frequencies of selection for the RND thinnings, based on an estimated planting density of 1750 trees ha^{-1} (e.g., six out of seven was the frequency of selection of trees to be felled in the Heavy thinning). The choice of trees to be felled was randomly determined and printed as numerous different binary sequences (1: preserve; 0: mark and fell), which were applied following the plantation rows. Empty positions (gaps) were flexibly accounted for as zeros to prevent overthinning. REG thinnings were constrained by the spacing between plantation rows (about 2.25 m). Every other row of trees was completely removed for the M-REG and H-REG thinnings, and the mean spacing between trees to be preserved in the remaining rows (all rows in L-REG) was calculated to obtain the desired tree density. Tree positions in adjacent rows were staggered to achieve an approximately diamond pattern. When several trees were close together, the smaller ones were preferentially marked to be felled, in accordance with wildfire prevention requisites. This selection added a “from-below” hint to both thinning methods but barely affected the spatial distribution pursued.

Once the thinning was completed in June 2005, three monitoring plots were established in each of the seven treatments (six thinning regimes + one control outside the fuelbreak). Plots were distant within each treatment

to ensure they could be considered independent representative samples, and were always located in areas which had a dense tree cover before thinning, so as to make the different treatments comparable by minimising heterogeneity among them. This approach was preferred over a standard design in blocks, which was less suitable for the creation of a fuelbreak and offered few advantages for our experimental conditions. Indeed, local heterogeneity in the forest (indicated by clumps of dead or slow growing trees) prevented having the desirable large homogeneous blocks where all experimental treatments can be tested under similar conditions. Some readers might consider this experimental design does not ensure that plots are real replicates, and should therefore disregard statistical results.

Tree growth and stand structure parameters

All plots ($n=21$) were defined by their central point, and included the nearest 25 trees. The DBH of the 525 plot trees was measured in March 2006, 2008 and 2010, before the beginning of the growth season. Additionally, the distances from each of these trees to their three nearest neighbours were taken using a measuring tape (1 cm precision) in March 2006. To avoid border effects, neighbouring trees were allowed to be outside the plots, which added 243 buffer trees to the sample. The DBH of the buffer trees was measured in March 2006 and March 2010.

Plot parameters

The surface area of plots (S) was calculated as the mean of the areas of two circles whose radii were the distances from the plot centre to the 25th and 26th nearest trees, and was employed to estimate tree density (λ , trees ha^{-1}). The DBH of plot trees was used to calculate stand basal area (G , $\text{m}^2 \text{ha}^{-1}$) and introduced in the regression equations to estimate the trunk, branch, fine fraction and total above-ground stand biomass (B , kg ha^{-1}). Thinning regimes were compared in terms of the growth observed in the stand basal area (ΔG) and stand biomass (ΔB) in the periods 2006-2008, 2008-2010 and 2006-2010.

The spatial distribution of trees in the plots was characterised using the Clark-Evans index (CE) (Clark and Evans, 1954). Through this index, the observed distances ($D_{i,j}$) between each plot tree (i) and its nearest

neighbour (j) were averaged and compared to the mean distance expected if trees were randomly positioned, which is a function of tree density (λ):

$$CE = \frac{\frac{1}{n} \sum_{i=1}^{i=n} D_{i-j}}{1/2 \sqrt{\lambda}}$$

To test whether the CE value of a plot significantly deviated from a random (Poisson) distribution ($CE = 1$), the normally distributed T_R test (Clark and Evans, 1954) was applied:

$$T_R = \frac{CE - 1}{\sigma} \quad \text{with} \quad \sigma = \frac{0.26136}{\sqrt{n^2/S}}$$

where σ is the observed variance and is calculated as a function of the number of trees in the plot (n , which equals 25 in our case) and plot surface area (S).

Individual tree parameters

Individual tree growth between 2006 and 2010 was calculated using Pressler's formula (Philip, 1994) for the periodic annual increment (PAI) in DBH , which expresses the mean annual DBH growth rate as a percentage of the mean DBH of the period:

$$PAI = \frac{1}{4} \frac{DBH_{2010} - DBH_{2006}}{(DBH_{2010} + DBH_{2006})/2} \times 100$$

Individual tree size in 2006 and 2010 (DBH_{2006} and DBH_{2010}), and growth (PAI) were analysed using the full plot-tree and buffer-tree samples ($n = 768$). DBH values from trees that died in the period ($n = 4$) were discarded and, in the tree-growth analysis, very small trees ($DBH < 10$ mm in 2006, $n = 9$) were also removed from the dataset, as their PAI could not be calculated or created very high outlying PAI values.

The data on neighbouring trees was employed to measure differentiation and competition among trees. Gadow's differentiation index ($TD3$; Gadow, 1993) was calculated for the 2006 and 2010 datasets using the DBH of each plot tree and those of its three nearest neighbours:

$$SCI3 = -\frac{1}{3} \sum_{j=1}^{j=3} \left(\frac{DBH_{\min i,j}}{DBH_{\max i,j}} \right)$$

where $DBH_{\min i,j}$ and $DBH_{\max i,j}$ are the smaller and the larger DBH in the pair-wise comparisons between a plot tree (i) and its j^{th} nearest neighbour.

A modified Hegyi's spatial competition index (Hegyi, 1974) was calculated for the 2006 dataset to measure the competition exerted on a plot tree by its three nea-

rest neighbours. This index (*SCI3*) compares the *DBH* of a tree (*i*) to that of its neighbours (*j*), and weights this ratio with the distance between them (D_{i-j} , in m):

$$SCI3_i = \sum_{j=1}^{j=3} \frac{DBH_j}{DBH_i * D_{i-j}}$$

TD3 and *SCI3* were calculated for 511 plot trees, after discarding the trees that were very small, died in the period, or had a neighbouring tree which died.

Statistical analysis

One-way ANOVA, with the seven treatments as fixed factors, and post-hoc tests were applied to determine which thinning regimes were different from

the Control and from each other. We had also planned to perform a two-way ANOVA with thinning method and thinning intensity as fixed factors. However, this analysis was discarded after detecting that Random and Regular thinnings of similar intensity differed notably in the residual tree densities obtained (see Fig. 1a).

In the analysis of plot parameters, where three repetitions (plots) were available for each treatment, the normality in the distribution of the data was tested for using the Shapiro-Wilk W test. Regarding individual tree parameters, the high number of independent samples ($n > 511$ trees) ensured that the sampling distribution of the mean approximated the normal distribution. Tukey's HSD post-hoc tests (Tk) were applied for

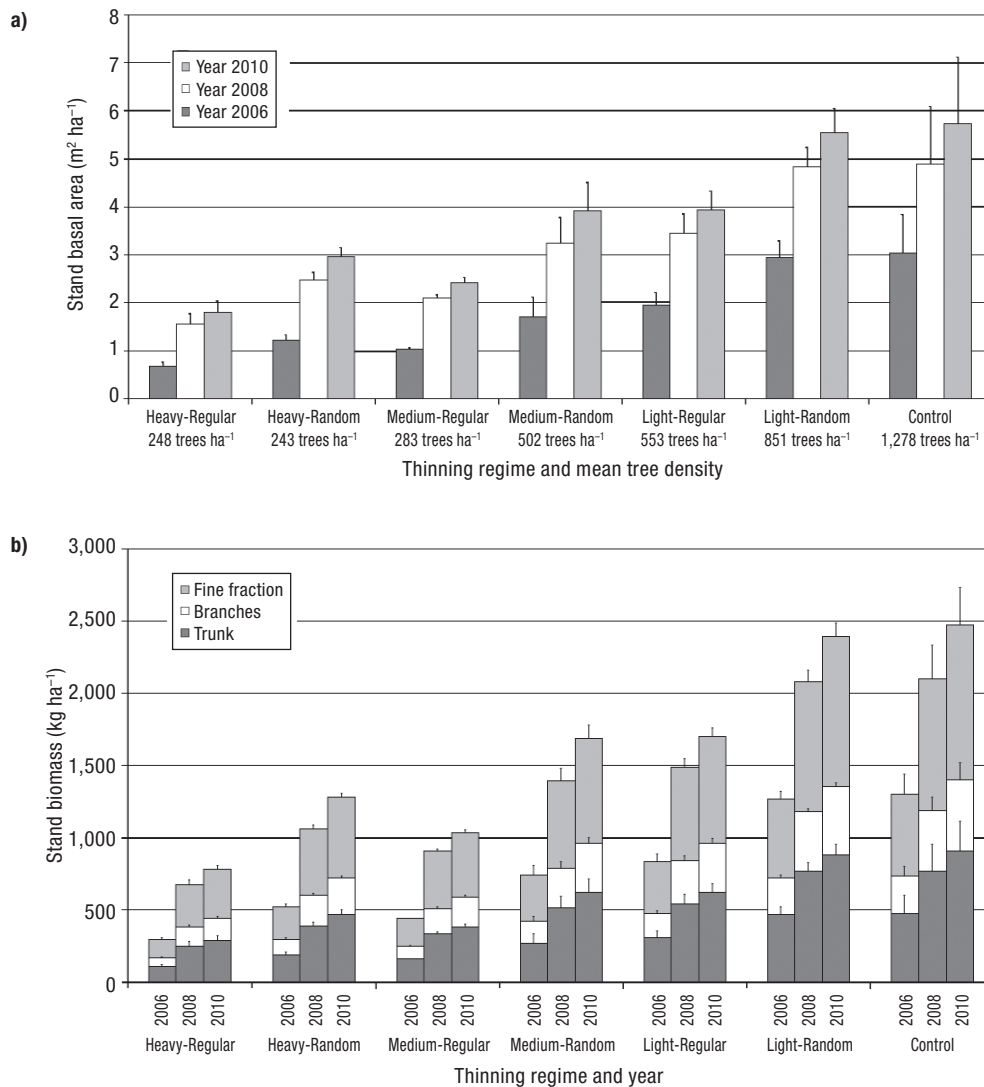


Figure 1. Evolution of a) Stand basal area and b) Stand biomass mean values in the different thinning regimes. Whisker lines indicate the standard error of the mean.

multiple comparisons when the homoscedasticity condition was met, which was verified using Levene's test. SCI3 data were log-transformed to meet this requirement. For other parameters with unequal variance among groups, Games-Howell post-hoc tests (G-H) were used instead. In all cases, a 95% confidence level was used to establish statistically significant differences. The SPSS 17.0 software (IBM-SPSS Inc., Chicago, IL, USA) was employed for the statistical analyses.

Results

Regression equations

The model selected for the regression equations of all fractions was $Biomass = a + b * DBH^2$. Accuracy was only slightly ($\Delta R^2 < 0.05$) increased by a model including also tree height [$Biomass = a + b * (DBH^2 * Height)^c$], so the more simple model was retained. The regression equations obtained jointly to meet the additivity property are shown in Table 1. Predicted biomass values should be considered less accurate for Aleppo pines with a size outside the range of the 32 sample trees ($1.2 \text{ cm} < DBH < 10.9 \text{ cm}$; $2.1 \text{ m} < \text{Total height} < 5.4 \text{ m}$).

Tree growth and stand structure parameters

Plot parameters

The mean tree density obtained after thinning was somewhat higher than desired in the M-RND and L-RND thinning regimes, while it was lower than intended in M-REG and L-REG (Fig. 1a). As a consequence, mean stand basal area and biomass values (Fig. 1b) for the latter two thinning regimes were similar to those

Table 1. Regression equations: Coefficients, Root Mean Square Error (RMSE) and Adjusted R^2 values (Adj- R^2) obtained in the estimations of above-ground biomass (kg) using the diameter at breast height (DBH, in cm). Biomass was divided in fractions according to its diameter (d). Model: $Biomass = a * DBH^2$

Biomass fraction	a coefficient	RMSE	Adj- R^2
Trunk ($d > 4 \text{ cm}$)	0.1245	1.029	0.929
Branches ($4 \text{ cm} > d > 1 \text{ cm}$)	0.0673	0.933	0.860
Fine fraction ($d < 1 \text{ cm}$)	0.1467	1.735	0.827
Total above-ground	0.3385	3.140	0.908

obtained for the Heavy thinnings and the M-RND thinning, respectively. After thinning, the stand basal area in the H-REG, M-REG and H-RND regimes fell by two thirds compared to the Control, while in the M-RND and L-REG plots, it was reduced by one third (Fig. 1a). On the other hand, the L-RND treatment obtained very similar values to the Control. The growth observed in the stand basal area (ΔG) was similar in all periods for the Control and for four of the thinning regimes (the three Random thinnings and L-REG). As for the other regimes, the ΔG s in H-REG and M-REG were statistically smaller than the growth observed in the L-RND thinning in 2006-2008 (G-H) and in the Control in 2006-2010 (Tk).

As stand biomass was estimated from the DBH of plot trees, the reductions observed after thinning were similar to those observed in stand basal area (Fig. 1b). Regarding stand biomass growth (ΔB), the H-REG and the M-REG thinnings were also those with lower values. Indeed, their ΔB for all fractions was significantly lower than in L-RND between 2006 and 2008 (G-H), and lower than in H-RND between 2008 and 2010 (G-H). Stand biomass growth over the total period 2006-2010 was also statistically similar in most groups, with the exception again of H-REG, which had a lower ΔB in all biomass fractions than the Control and the L-RND thinning (Tk), and M-REG, which grew less than the Control (Tk).

By 2010, total above-ground biomass in the heavier thinnings (H-REG, H-RND and M-REG) had not reached the mean values observed in the Control plots in 2006 ($13,000 \text{ kg ha}^{-1}$). The mean amount of fine fraction biomass grew from $1,800$ to $4,500 \text{ kg ha}^{-1}$ in the heavier thinnings between 2006 and 2010. This constituted 32 to 42% of the amount of fine fraction biomass estimated for the Control plots, which grew from $5,600$ to $10,700 \text{ kg ha}^{-1}$.

The Clark-Evans index values were lower (Tk) in H-RND than in all the other thinning regimes (Fig. 2). The T_R test showed that the distribution of trees in all three H-RND plots could be considered random. The rest of the plots tended to regular tree spacing and statistically differed (most with $p < 0.001$) from a random distribution.

Individual tree parameters

DBH 2006 was significantly larger in all thinning regimes than in the Control, the contrast being particularly remarkable for H-RND (Fig. 3a). Tree growth

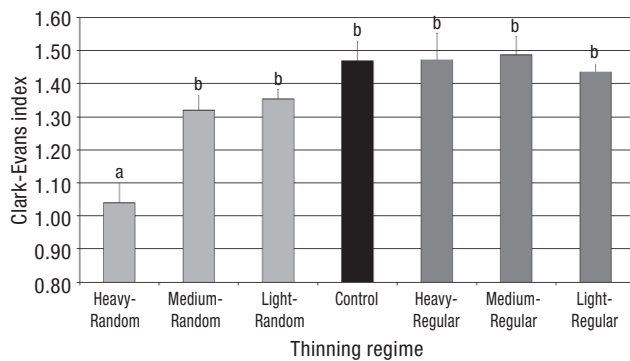


Figure 2. Tree spatial distribution: mean Clark-Evans index values in the different thinning regimes. Whisker lines indicate the standard error of the mean. Different letters indicate statistical differences among thinning regimes according to Tukey's HSD test.

(PAI) in the following four years was also greater in the thinned areas, and especially in the heaviest thinnings. As a result, mean DBH 2010 across thinning regimes was notably (between 21% in L-RND and 68% in H-RND) larger than in the Control area.

Gadow's DBH differentiation index (TD3) was very similar across all thinning regimes in 2006 and 2010 (Fig. 3b). Differences with the Control were accentuated by 2010, after a general decrease in TD3 values which was particularly acute in the areas which had undergone more intense thinnings. The greatest contrast was registered in 2010 between the H-RND thinning (TD3 = 0.20) and the Control (TD3 = 0.30). These values indicated that, on average, the smallest tree in a neighbouring pair had a DBH of 80% (70%) of the

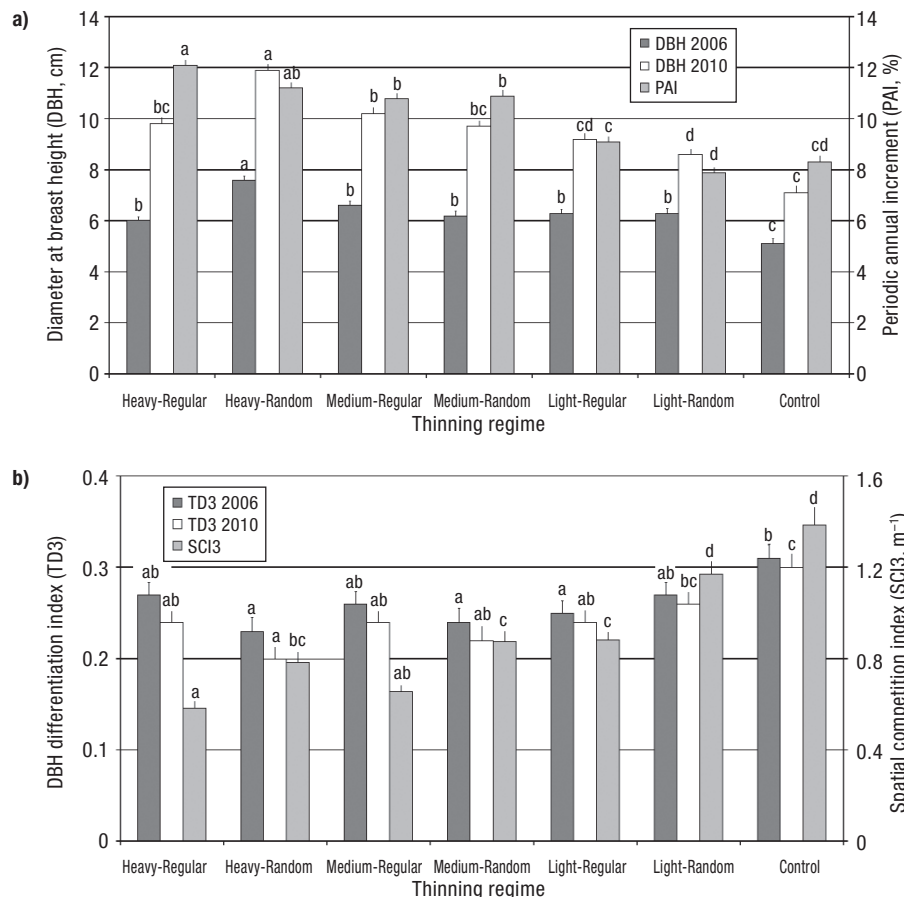


Figure 3. a) Tree size and growth: diameter at breast height (DBH) in 2006 and 2010, and periodic annual increment (PAI) mean values in the different thinning regimes; and b) Tree size differentiation and competition: Gadow's diameter at breast height (DBH) differentiation index (TD3) in 2006 and 2010, and Hegyi's spatial competition index (SCI3) mean values in the different thinning regimes. Different letters within a parameter series indicate significant differences ($p < 0.05$) among groups according to Tukey's HSD (DBH 2010, PAI, TD3 and SCI3) or Games-Howell's (DBH 2006) tests. SCI3 data were log-transformed before the statistical analysis.

largest one in the H-RND thinning (Control). All thinning regimes, except L-RND, reduced tree competition significantly, as they had statistically smaller SCI3 values than the Control (Fig. 3b). SCI3 tended to be greater in the Random, as well as in the less intense thinnings.

Discussion

The experimental thinning regimes described in this study produced remarkable differences in tree size and growth, as well as in the stand structure of a young Aleppo pine MPCF. In accordance with our hypotheses, heavier thinnings brought about greater changes than seen in the Light intensity regimes, which were similar to the Control in several parameters. Random and Regular thinnings differed in their effects on stand structure, particularly regarding the spatial distribution of trees. Tree growth tended to be lower in Random thinnings, but our results were not conclusive, as this parameter was affected to a large extent by the thinning intensity, which was unexpectedly heavier among Regular thinnings.

All Random thinnings produced some randomisation in the spatial distribution of trees, as measured by the Clark-Evans index (Fig. 2). However, it was in the H-RND thinning ($CE = 1.04$) that the initial regularity of the planting pattern was fully transformed, which was also noticeable on-site. This thinning regime can, thus, be recommended when an effective randomisation in the spatial distribution of trees is pursued in a young MPCF. This regime would lead to a low density stand with long-lasting gaps in the canopy, as the growth of the irregularly spaced remaining trees is not able, at least in the short term, to cover the larger gaps associated to a random tree distribution. According to Montero *et al.* (2001), this spatial structure resembles that of natural Aleppo pine stands. For fuelbreaks, these long-lasting gaps contribute to maintaining the horizontal discontinuity in canopy fuels for longer, reducing the risk of a difficult-to-fight crown fire reaching the firebreak (Agee *et al.*, 2000). Nevertheless, maximum canopy cover is recommended for fuelbreaks where shading the undergrowth is necessary to prevent surface fuel accumulation (Vélez, 2009). Regular thinning methods could better suit this latter requirement.

No previous references to a random method of thinning have been found, although other methods may produce

similar effects. In a combined systematic and selection (from below) thinning on a *Pinus sylvestris* plantation, for instance, thinned stands also tended towards a random tree distribution, particularly in the heavier thinning ($CE = 1.11$) (Crecente-Campo *et al.*, 2009). In contrast, in a long-term thinning experiment in north-east China, thinning from below was found to produce some departures from the random distribution of trees observed in control plots (Lei *et al.*, 2007). In a post-fire regenerated Aleppo pine stand in south-eastern Spain, early thinning was found to increase the initially low CE values (0.14-0.68) and promote a non-clustered tree distribution (Moya *et al.*, 2009). Further research using other thinning methods in MPCF could provide a useful contrast to the results we obtained in tree distribution through random thinning.

The divergence observed between the desired tree density and that obtained through the thinning (Fig. 1a) was probably due to the tree marking and felling procedure used. Greater densities after Random thinnings may be attributed to an underestimation of the initial tree-planting density, as this density was directly related to the calculated frequency of tree selection. Lower densities after Regular thinnings were most likely caused by missing trees along plantation rows. We underestimated their potential effect on final tree density, and as quantifying it *a priori* was not feasible, no correction was introduced in the calculation of the mean spacing between residual trees.

Stand basal area and biomass growth was larger in the first two years after thinning than in the 2008-2010 period (Fig. 1a and 1b). Rather than a short-lived effect of thinning, this was most likely due to climatic variability or, rather, to the severe infestation by the pine processionary moth that affected the whole study area from the end of 2007 onwards. The fact that growth was also reduced in the non-thinned Control area, as well as in the more heavily thinned areas, which could be expected to benefit from the competition release for longer, corroborates the very limiting effect of the pest on tree growth. This ill-timed event could have also minimised the contrasts in tree growth among thinning regimes.

The heaviest thinnings we applied (H-REG, M-REG and H-RND, in this order) caused a drastic diminution in stand basal area, which conditioned the growth of this parameter and of stand biomass in the four years after thinning (Fig. 1a and 1b). An understocked stand is a known consequence of heavy thinnings (Del Río *et al.*, 2008b; Crecente-Campo *et al.*, 2009) and may

not be desirable when maximum timber production or carbon sequestration are the main management goals (Del Río *et al.*, 2008a). In these cases, the Light thinning intensity could be considered more suitable. From a wildfire prevention perspective, though, understocking could help lower biomass (fuel) accumulation in trees. Focusing on the (most fire-prone) fine fraction, the amount of biomass estimated in the heaviest thinnings (just over one third of the Control) implied that the risk of crown fires was markedly reduced through thinning. Indeed, by 2010, the fine fraction biomass in Control plots had reached the usual range (9,600–18,000 kg ha⁻¹) observed in adult Aleppo pine forests (Mitsopoulos and Dimitrakopoulos, 2007), posing a high crown-fire risk.

From a biodiversity perspective, the thinnings we applied increased the herbaceous species richness in the understory when compared to the Control (Navarro *et al.*, 2010). The chances of native shrubs to grow under the pine canopy in a MPCF may, however, be somewhat limited in areas of low rainfall (Osem *et al.*, 2009). Indeed, Aleppo pine afforestations have been found to negatively affect the presence of native shrubs (Chirino *et al.*, 2006), and particularly late-successional species (Maestre and Cortina, 2004). Thinnings could mitigate some of the negative effects of a dense MPCF (*e.g.*, the depletion of soil moisture) although more specific measures may be required, particularly in low rainfall areas, if the goal is to promote a species-rich forest.

All thinning regimes resulted in a greater mean DBH 2006 than the Control (Fig. 3a), and this can be attributed to the “from below” selection superposed on the applied thinning methods. This tendency was particularly acute in the H-RND thinning, as the smaller number of residual trees and the lack of spatial restriction allowed greater margin to select larger individuals. The subsequent growth amplified the differences with the Control, leading to significantly larger trees in all thinning regimes. In fact, the increased growth after thinning (which was completed in June 2005) might have even affected somewhat the DBH 2006 measurement. From a wildfire perspective, larger trees are an asset for forest resistance, as trees with thicker bark (that insulates inner tissues from heat) and higher position of the foliage (which prevents crown scorching) can be expected to better withstand low intensity wildfires (Fernandes *et al.*, 2008). Besides this, early thinning has been found to promote cone production (González-Ochoa *et al.*, 2004; Verkaik and Espelta,

2006), which is particularly relevant for the resilience of young dense stands. In the event of wildfire, these stands would more likely be completely burnt and require serotinous cones to regenerate.

Rather than to type of thinning, tree growth seemed to be more strongly related to thinning intensity, to the extent that the two Light thinning regimes were found to be similar to the Control (Fig. 3a). The values obtained for the modified Hegyi's spatial competition index (SCI3, Fig. 3b), an index previously found to be correlated to tree growth (Crecente-Campo *et al.*, 2009), suggest that greater growths were stimulated by lower degrees of competition with the neighbouring trees. In fact, SCI3 was lowest in the H-REG thinning regime, a combination of low tree density and regular spacing that maximised individual tree growth.

Thinning tended to reduce DBH differentiation among neighbouring trees, as measured by TD3 (Fig. 3b). This is likely a result of the “from-below” approach that characterised all thinning regimes, as many of the smallest trees were felled, and this homogenised tree sizes. This consequence may be considered undesirable from a diversity perspective, but felling the lowest layer of trees is frequently regarded as necessary to increase the height of the live crown and eliminate ladder fuels (Graham *et al.*, 1999; Agee *et al.*, 2000; Vélez, 2009). This preventive approach may be crucial in Aleppo pine forests, as they usually have a low crown height (3–6.5 m) which increases the risk of crown fire initiation even under moderate burning conditions (Mitsopoulos and Dimitrakopoulos, 2007).

However, differences in TD3 among thinning regimes were not very notable, as other authors have also reported (Moya *et al.*, 2009). TD3 decreased with time in all cases, in agreement with previous references (Crecente-Campo *et al.*, 2009), but the decrease we observed was greater in heavier thinnings. This is likely to have been an effect of a greater competition release in those regimes, which would have facilitated the growth of the previously dominated smaller trees. If this trend continued, the more heavily thinned areas would have a remarkably lower local differentiation in DBH in the future.

Conclusions

Overall, each of the six thinning regimes studied produced distinct effects on tree size and growth, as well as in the stand structure of the experimental Aleppo

pine planted forest. The appropriate choice and adaptation of the thinning regime to be used will depend on the MPCF management goals. According to our results, a regular plantation pattern can successfully be transformed into a random tree spatial distribution through heavy random thinning. If heavy thinning is undesirable, it may be worth carrying out a lower intensity random thinning, but the degree of wildfire prevention attained will be limited and further thinning will be necessary in the future to accomplish a random tree distribution.

Regular thinnings tended to increase individual tree growth and, applied at a light intensity, may be the most suitable approach to maintain competition and prevent development of the undergrowth when this stratum poses a high fire risk. This thinning intensity will also avoid understocking the forest stand, which ensures higher timber production and carbon sequestration, but improvements in diversity will probably be limited. Indeed, all thinning regimes resulted in lower differentiation in tree size, which may not be desirable from a diversity perspective. Nevertheless, given the limited resources available for MPCF management, thinning is most likely to be adopted on a small scale, such as when it is applied to create a fuelbreak. Such localised actions will unavoidably contribute to increasing the diversity of stand structures harboured in the larger MPCF where they are applied.

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