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# FUEL CHARACTERIZATION AND EFFECTS OF WILDFIRE RECURRENCE ON VEGETATION STRUCTURE ON LIMESTONE SOILS IN SOUTHEASTERN FRANCE.

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## Abstract

Recurrent fires are more frequent in the Mediterranean basin due to global climate change. The effects of fire recurrence (up to more than three recurrent fires with an interval since the last fire of more or less than fifteen years) on the structure of fuel beds were studied at stand level in southeastern France. Three types of vegetation that are among the most common in the study area (pure pine stands, oak-pine mixed stands and shrublands) were studied to determine the role of fire recurrence on biodiversity. Fuel bed description was carried out on aboveground vegetation at understorey and overstorey levels, and on litter in 39 homogeneous study sites. The structure of the understorey was categorized following a gradient of fragmentation including: (i) isolated shrubs, (ii) vegetation patches made up of a single species and (iii) homogeneous vegetation layers including different species of the same height.

The decrease of time interval since the last fire caused an opening of the canopy, except for the lowest vegetation stratum (grasses and low shrubs) that did not vary significantly. There was no significant change in horizontal connectivity between fuel beds (% of layer cover) as a function of the fire regime or the type of vegetation. Conversely, the vertical connectivity (% of tree cover, height of the trees and of the layer) was affected by the fire regime, especially in

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pure pine stands. Fire regime also influenced the litter layer and litter depth decreased with an increase in the opening of the stand that could be due to fire. These results show that fire risk is greater in the oldest stands (long interval since the last fire).

*Key words*: wildfire recurrence; fire interval, wildland fuel, vegetation structure; shrubland, *Pinus halepensis*

## Introduction

In recent decades, the number of fires has increased in most regions of the Mediterranean basin (Moreno *et al.,* 1998; JRC, 2006). Humans often play an important role in both temporal and spatial patterns of fire occurrence and over 95 % of current fires in the South of France are ignited by people (Long *et al.,* 2008). The incidence of fire is creating a new ecological situation that may enhance desertification processes (Vallejo, 1997) due to global climate change. As Naveh (1990) pointed out, fire frequency can determine the structure of ecosystems. Going beyond the optimum cycle of disturbances may cause the disruption of a number of basic ecosystem properties such as the pool of soil nutrients. According to Eugenio *et al.* (2006), fire recurrence is defined as the number of fire events occurring at a site during a given period of time. Many studies have only taken into account the effect of the number of fires (Gondelis *et al.,* 2008; Eugenio and Lloret, 2006; Eugenio *et al.,* 2006; Pausas and Lloret, 2007; Kazanis and Arianoutsou, 1996; Clemente *et al.,* 1996). However, the time interval between two successive fires can also have different impacts on the vegetation. Donato *et al.* (2008) showed that much of the native biota was resilient to “extreme” events such as recurrent severe fires. Conversely, short intervals (e.g. less than 4 years in Spanish Catalonia ; Diaz-Delgado *et al.,* 2001) may prevent seed-producing species from replenishing seed banks and resprouters species from replenishing underground reserves (Canadell and Lopez-Soria, 1998; Keeley *et al.,* 1999). High fire recurrence (i.e. short time intervals) may

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also have deleterious effects on soils causing nutrient deficits that affect plant growth and primary production (Carter and Foster, 2004; Ferran *et al.,* 2005). Fire intensity is also an important parameter to consider because it influences post-fire regeneration in burned areas (Agee, 1998). In spite of enhanced suppression efforts in most Mediterranean countries, mainly using clear-cutting or sometimes prescribed burning, high-intensity large fires are still common (Moreno *et al*., 1998; Keeley *et al.,* 1999). Prescribed burnings are supposed to decrease the intensity of a subsequent wildfire by disrupting the horizontal and vertical continuity of the fuel complex, but their hazard reduction potential is not easily generalized (Fernandes and Botelho, 2003). Moreover, the effects of fire on vegetation structure and on the resulting changes in the fire regime at large spatial and temporal scales are poorly understood.

Studies on the impact of fire have been carried out at different levels, for example by Fernandes *et al.* (2008) on the impact of fire severity at tree level. The impact of fire recurrence (especially the number of fires) and of fire severity have frequently been analyzed at stand and landscape level. Variations in fire severity are a potentially important source of diversity that affects vegetation patterns at landscape scale (Christensen, 1993; Chappell and Agee, 1996; Taylor and Skinner, 1998). Variable fire regimes may promote landscape heterogeneity, diversification in vegetation patterns and biotic diversity (Scholes and Walker, 1993). Many of the observed patterns in any landscape result from past disturbances and in turn, landscapes patterns can influence the spread of future disturbances (Turner *et al.,* 1989). Probabilistic models have been used to investigate, for example, interactions between landscape patterns and crown fires (Gardner *et al.,* 1987; Turner *et al.*, 1989) revealing the importance of landscape connectivity, and consequently the spatial arrangement of fuels, in controlling disturbance dynamics (Davis and Burrows, 1994; Turner and Romme, 1994).

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In this study, vegetation was analyzed as fuel, with the aim of predicting fire behaviour characteristics (rate of spread, intensity, flame height, etc.), as well as the impact of fire. Wildland fuel is any combustible material available for fire ignition and combustion (Keane *et al*., 2001), mainly live and dead organic matter. Fuels are inherently complex and have many properties that determine fire behaviour, which make them difficult to characterize (Sandberg *et al*., 2001).

Pine-dominated ecosystems are one of the main landscape features in the Mediterranean basin, where they account for 25% of forested surface (Barbéro *et al.*, 1998). The most abundant and widespread pine species in the western part of the basin is *Pinus halepensis*, which accounts fort 2.5 million ha (Quézel, 2000). This species is known to have expanded during the last century, both due to its widespread use for afforestation and to its ability to invade abandoned fields and pastures (Lepart and Debussche, 1992). In the French Mediterranean Provence, *P. halepensis* forms communities where it coexists with other tree species such as oaks (*Quercus ilex, Q. pubescens, Q. coccifera)* or *Arbutus unedo*. The forest understorey is dominated by shrubs including *Pistacia lentiscus*, *Phyllirea* sp. pl., *Rhamnus* sp. pl., *Ulex parviflorus*, *Quercus coccifera*, *Rosmarinus officinalis*, *Erica* sp. pl., and *Cistus* sp. Pl.. Such communities have high fuel loads and vertical fuel continuity, and promote high intensity fires that usually kill the entire pine population (Eugenio *et al.,* 2006).

The aim of this study was to address the effects of fire recurrence at stand level in southeastern France, especially the effect on vegetation structure of the time elapsed since the last fire. A further aim was to determine if these parameters affect forest biodiversity. This study was carried out on limestone soils at the same time as similar studies taking place on acidic soils in Provence, France.

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## Methods

Study sites

In order to prevent variability due to site fertility or physiography, data were collected on sites distributed throughout limestone-derived soils in Provence, southeastern France (Fig. 1) that were selected because of their homogeneity in terms of fertility (medium according to the forest site classification), elevation (less than 300 m), slope (less than 2°) and aspect. A total of 39 study sites were selected using geographical information system (GIS, Esri ArcGis 9.1) by intersecting the following layers: (1) an accurate vegetation map (National Forest Inventory, IFN), (2) fire boundaries for the period 1960[1]-2005, and (3) a Digital Elevation Model from the altimetric database provided by IGN (French National Geographical Institute). Within each study site, the vegetation was described in a representative 400 $m^2$ georeferenced plot (20 m x 20 m).

Figure 1

Vegetation types and fire regimes

Three representative vegetation types were sampled: pure *Pinus halepensis* stands, mixed pine-oak (pines with *Quercus ilex* and/or *Quercus pubescens*) stands, and shrublands (so-called Garrigue, i.e. evergreen sclerophyllous vegetation type dominated by shrubs). These three types are representative of three sequences of post-fire succession on limestone substrates in Provence and dominate in the study area.

Sites differed in their fire regime which was defined, first, according to the number of fire events that had occurred during the past 45 years (3 types: no fire, 1 or 2 fires, 3 or more

[1] No spatial records are available on fires occurring before 1960 in the study area.

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fires), and second, according to the time interval since the last fire (3 types: more than 45 years with no fire, between 45 and 15 years and 15 years or less) (Table 1). According to Eugenio *et al.* (2006), a period of at least 15 years without a fire appears to be very important for the recovery of *Pinus halepensis* populations. This interval is assumed to be sufficient to favour forest succession towards oak woodlands (Gracia *et al*., 2002). The fire regime "no fire for 45 years" was used as control.

Description of vegetation structure

Study plots were marked with differential GPS sub-metre post-processed positioning using EGNOS (Gauthier *et al.*, 2003). Fuel structural elements were referenced according to the geographical coordinates of the plot using decameters. The aboveground vegetation and litter of each plot were described:

- Aboveground vegetation

Aboveground vegetation strata were assumed to be composed of live fuel. The stand description took three structural elements into account: (i) isolated trees or shrubs, (ii) vegetation patches made up of a single species and (iii) continuous fuel beds in the understorey vegetation, including different species of the same height, defined as a "vegetation layer" Therefore, the variation of structures was also assumed to be part of landscape diversity.

The following parameters were recorded for each type of vegetation structure:

(i) Isolated shrubs and trees: height (m) of the first live branch and trunk circumference (m) at 1.30 m height giving the diameter at breast height (dbh) only for the trees, crown diameter measured on both North-South and East-West axes (m), total height (m) of the plant and its species,

(ii) Vegetation patches: patch diameter measured on both North-South and East-West axes (m), mean patch height (m) and its species,

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(iii) Vegetation layers: the vegetation was sampled along parallel transects laid out every three metres, perpendicularly to the side of the plot, within the continuous fuel bed. Every three metres along each transect, the main species, its height (m) and its percentage cover per 1 $m^2$ were recorded.

Isolated shrubs, patches and layers generally composed the understorey in the forest. The tree density (number of trees per hectare) and the volume of tree crowns ($m^3 \cdot ha^{-1}$), the percentage cover of the different heights of the vegetation strata (0-1 m, 1-3 m, 3-6 m, 6-10 m, >10 m), the percentage cover and mean height (m) of each structural element (tree, shrub, patch and layer) were also calculated.

Litter

Litter is the ground fuel layer made up of dead vegetation which is more or less decomposed (leaves, needles, twigs, etc.). All measurements were made in April-July, just after the peak of leaf/needle fall and before their decomposition, which occurs in autumn-winter in the study area. Litter depth (cm) and percentage cover were recorded in 1 $m^2$ quadrats, placed systematically every two metres (Fig. 2). The average of the values gave the mean depth and mean cover for each plot.

Figure 2

Statistical analyses

The effects of both factors “type of vegetation” and “fire regimes” on the fuel structure variables were analyzed using one-way analysis of variance (one-way ANOVA). In addition to tests of overall significance with ANOVA, the LSD test was used to test significant differences between vegetation types or fire regimes. The α level for all tests was 0.05. In all cases, when the distribution of data did not follow the expected parametric pattern and when the log- or square root-transformations were not efficient, a non-parametric Kruskal-Wallis

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test was used instead of a parametric ANOVA. Then, for each type of vegetation, different fire regimes were compared with respect to the cover of vegetation strata and of the fuel structural elements, using a non-parametric Kruskal-Wallis test because of the small number of data in each sample analyzed. The relationship between litter variables was analyzed using regression and the Pearson correlation coefficient. A significant relationship between the variables was assumed when the probability was less than 0.05 (Statgraphics Centurion XV).

## Results

- Aboveground vegetation

The following vegetation structure parameters differed significantly according to the types of vegetation (Tab. 2 and 4):

(i) Height of shrubs, vegetation patches and layer. The lowest values were recorded in shrublands and the highest in pine-oak mixed stands. Tree height did not vary significantly between the different stands.

(ii) Cover of vegetation strata, except for the 1-3 m stratum which did not vary from one type to another. The cover of the lowest stratum (0-1m) increased with a decrease in the canopy cover, unlike the cover of the highest strata which displayed the reverse trend.

(iii) Cover of trees, shrubs and vegetation patches which decreased more in more open environments, i.e. in shrublands. The cover of the vegetation layer did not vary from one ecosystem to another.

(iv) Tree density and volume of tree crowns per hectare. The highest values were obtained in *Pinus halepensis* stands.

The fire regimes entailed the following significant changes in vegetation structure (Tables 3 and 4):

(i) A decrease in the height of the trees and of the vegetation layer with fire. Patch height remained constant regardless of the fire regime. Shrub height only varied significantly with the number of fires.

(ii) A decrease in strata cover, especially in the case of recent fires (in the 3-6 m and 6-10 m strata), except in the lowest stratum (0-1 m) in which the cover remained constant regardless of the fire regime.

(iii) A decrease in tree cover with fire, while the other types of vegetation remained constant.

(iv) A decrease in tree density and in the volume of tree crowns with fire.

<u>Effect of fire regime on vegetation structure within each vegetation type</u>

The previous results, especially those concerning the impact of vegetation types and fire regimes on the cover of vegetation strata and of fuel structural elements were refined by comparing each fire regime within each vegetation type. These two variables were considered to be the most significant in describing the forest diversity at landscape level.

Thus, in shrublands, there was no variation in the stratum cover according to the fire regime except in the 1-3 m stratum in which the cover was significantly higher (KW=9.48, p=0.023) in unburned stands or in stands that burned a long time ago (1-2 fires > 15 years ago) than in stands that burned recently after a short interval (1-2 fires ≤15years ago and ≥3 fires <15years ago).

The same trend was found in mixed stands in the upper strata, 3-6 m and 6-10 m (KW=8.06, p=0.018 and KW=5.6, p=0.05). The cover of the highest stratum (>10 m) was significantly higher in unburned stands (40%) than in burned stands (0%) regardless of the number of fires and the interval since the last fire (KW=10.5, p=0.005). There was no variation in the cover of the lower strata (0-1 m and 1-3 m) as a function of the fire regime.

In *Pinus halepensis* stands, the impact of the fire regime on the cover of strata was significant only in the highest stratum (>10m), and there was a marked decrease in cover from unburned stands to burned stands (KW=6.32, p=0.042) (Table 5; Fig. 3).

Concerning the fuel structural elements cover, our results showed that, in shrublands, the impact of the fire regime was not significant regardless of the structural element concerned (isolated trees or shrubs, patches, layers). In mixed and pure pine stands, the impact of the fire regime was significant only for tree cover (KW=10.35, p=0.006 for mixed stands and KW=8.44, p=0.014 for pine stands) with a decrease in tree cover from unburned or previously burned stands (> 15 years) to recently burned stands (≤15years) (Table 5; Fig.4).

These results showed that, in shrublands, the fire regime did not lead to a variety of vegetation patterns. In both pure pine and mixed stands, the variation in vegetation structure due to fire could involve a variation of landscapes only at tree level. As the sampling was only carried out on 39 study sites, all of which were representative of limestone Provence, this could explain the lack of variation in our results.

Figures 3 and 4

Figure 5 shows the cover of the main species recorded on the study sites according to vegetation types and fire regime. The minimum number of main species was observed in pure pine stands that had never burned (2 species) and in shrublands that had burned repeatedly. The maximum (4 or 5 species) was observed in the three types that had burned once or twice with a long interval since the last fire (>15 years) and in mixed stands that had burned once or twice with a short interval (≤15 years) since the last fire.

Figure 5

- Litter

Results showed that both fire the regime and the vegetation type had a significant influence on litter depth (ANOVA, F=6.17, p=0.002; F=8.17, p=0.0013) whereas only the fire regime had a significant influence on the cover of litter (ANOVA, F=5.08, p=0.0053). The lowest values of these variables were linked to the highest number of fires and to open land (shrublands). An LSD test showed that differences were significant only between shrublands and mixed stands, and between shrublands and *Pinus* stands for vegetation types. Concerning the fire regime, differences were significant between the oldest fires (0 fire since 1960 and 1 or 2 fires > 15 years) and recent fires (<15 years regardless of the number of fires). There was a significant decrease in litter depth and cover with an increase in the number of fires and with a decrease in the interval since the last fire, especially for shrublands (Fig. 6).

There was a moderate positive correlation between litter depth and the cover of the 1-3 m stratum (correlation coefficient=0.53, p=0.02), and the cover of the 6-10 m stratum (correlation coefficient=0.52, p=0.02). The correlation was the same between the cover of the litter and the cover of the 1-3 m stratum (correlation coefficient=0.55, p=0.014). There was no correlation between litter parameters and tree basal area or between litter parameters and tree cover.

Figure 6

## Discussion

Change in vegetation structure

The results of our study showed that the minimum cover of trees, shrubs and patches was observed in shrublands, early stages in the post-fire succession (Quézel, 1998), and that coverage increased with stand aging. In the both types of forest, tree cover decreased with a decrease in the time since the last fire (i.e. short time interval since the last fire), as no trees or a few trees were recorded in the case of a recent fire (≥3 fires or 1-2 recent fires and ≤15 years). This was especially true in the case of the cover of the highest stratum (>10 m) in *P. halepensis* stands, which were the most mature encountered in the study area. Indeed, several studies have shown that tree cover was the most relevant variable to explain differences in the characteristics of Mediterranean plant communities (Gondelis *et al.,* 2008; Eugenio and Lloret, 2006; Eugenio *et al.,* 2006; Pausas and Lloret, 2007; Kazanis and Arianoutsou, 1996; Clemente *et al.,* 1996). In these studies, fire intervals ranging from 2 to 10 years negatively affected most ecosystem attributes such as soil organic matter, and led to structural simplification of the vegetation. For example, in *Pinus halepensis* populations, the percentage of pines decreased while that of shrubs increased. In our study sites, the cover of understorey (shrubs, patches and layers) did not vary significantly between burned and unburned sites regardless of the type of vegetation (Table 5). Nevertheless, the layer was the main structural element (always more than 40% cover) allowing high horizontal connectivity between fuel beds, regardless of the vegetation type or the fire regime.

Tree cover, tree density and volume were the highest in pure pine stands in which *Pinus halepensis* had the largest crowns and dbh, but these values decreased with the recurrence of fires. This revealed a loss of resilience of the *P. halepensis* population with an increase in the number of fires and/or a decrease in the time interval since the last fire (Eugenio *et al.,* 2006). In recently burned sites, *P. halepensis* was dominant only in pure pine stands, but not in mixed stands. Several authors (Bradstock *et al.,* 1997; Pausas, 1999) showed that recurrent fires favoured seeders only if the fire interval enabled plant maturation and production of seeds.

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Generally, the cover of every stratum except the lowest (0-1 m) decreased with a decrease in the canopy cover. Their values were the lowest in shrublands and in the case of recent fires (short time interval since the last fire) and the highest in pine stands for the highest stratum (>10 m). The cover of the lowest vegetation stratum (0-1 m), made up of grasses and low shrubs, remained constant regardless of the fire regime and was highest in shrublands.

Some *Pinus* trees were present above the shrublands in one plot and in mixed stands. These trees were as high as those in pure pine stands, explaining why tree height did not vary according to the type of vegetation in the study site. Only tree height and layer height decreased in recent fires leading to a decrease in vertical connectivity. This decrease in tree growth also reflected loss of resilience as reported by Eugenio *et al* (2006). The height of understorey was lowest in shrublands and highest in mixed stands, which were more open than mature *Pinus halepensis* stands where the soil is too acidic to allow development of an important understorey (Barbier *et al.,* 2008). Vertical connectivity was higher in mixed stands than in pure pine stands.

In the study sites, shrublands were mainly composed of *Quercus coccifera* as was the understorey of *P. halepensis* stands (Figure 5) regardless of the fire regime. Indeed, this species regenerates after fire by resprouting (Deletti *et al.,* 2005) and has been shown to be very resilient to wildfire (Trabaud, 1990; Canellas and San Miguel, 1991; Papio, 1994) because of its extensive root system (Papatheodorous *et al.,* 1998; Canellas and San Miguel, 2000). Several authors showed that, after a period of a few years, plant species composition becomes similar to prefire composition, the most efficient being the resprouters, especially in shrublands (Vallejo *et al.,* 1998; Nunez and Raffaele, 2007; Delitti *et al.,* 2005). Our results also showed that *Quercus ilex*, another resprouting species, was the main understorey species in both unburned and recently burned plots in mixed stands (up to 73%) while *Ulex*

*parviflorus*, an obligate seeder, was one of the dominant understorey species in recently burned plots (up to 44%). This result differs from that of Delitti *et al*. (2005) who found that, on the most frequently burned sites, the cover of herbaceous and sub-shrub species showed a tendency to increase while that of the obligate seeding shrubs (*U. parviflorus*) decreased with both the frequency of fires and a decrease in the interval since the last fire. According to Clemente *et al.* (1996), obligate seeders have different patterns of recovery, some species being more abundant just after a fire (*Cistus monspelliensis, Cistus albidus* in the recent fires regimes), while others display delayed yet continuous recruitment during post-fire succession (*Rosmarinus officinalis* after old fires and the control). In the present study, for both fire regimes (old or recent fires), *Pinus halepensis* remained the main species in pure pine stands. *P. halepensis* is a seeder and a serotinuous species that forms a canopy seed bank (Goubitz *et al.*, 2003; Daskalakou and Thanos, 1996; ). Post-fire regeneration of the species relies exclusively on the canopy seed bank, due both to the transient nature of the soil seed bank and to the destruction by fire of all seeds on or near the soil surface (Habrouk *et al.*, 1999; Izhaki *et al.*, 2000; Núñez and Calvo, 2000).

Relationships between fuel accumulation and wildfire activity have been reported in Europe by Rego (1991). The management of the wildfire risk involves the control of the understorey growth, generally by clear-cutting or prescribed burning. The latter is generally used at wide scale even if the fuel accumulation rate frequently limits prescribed burning effectiveness to a short post-treatment period according to Fernandes and Botelho (2003).

Impact of fire recurrence on litter

Litter depth decreased in the case of recent fires as a function of the decrease in tree cover and height. Litter depth values were lowest in shrublands where species did not produce a large amount of litter whereas understorey productivity was the highest in open stands (Wimberly and Kennedy, 2008). The litter was thicker under the oldest stands (>15 years) than under

recently burned stands which had a greater tree cover. According to Cseresnyes *et al.* (2006), needlefall increased with stand age and Williams and Wardle (2007) showed that mean annual pine needlefall was 2.5 times greater in unburned areas than in burned areas. Indeed, they found a significant reduction in leaf shed in both species and a significant reduction in litterfall for up to four years following the fire.

Litter cover also decreased in the case of recent fires regardless of the type of vegetation. Williams and Wardle (2007) showed that the majority of pine litter (70%) fell directly below the pine canopy, whereas the proportion decreased in mixed stands.

The correlations between litter parameters and vegetation strata (1-3 m and 6-10 m) cover were intermediate. The 1-3 m stratum was one of the two strata with the highest cover especially in mixed stands (>55%). This stratum was essentially made up of tall shrubs with a cover of more than 30% regardless of the type of vegetation and the fire regime (except when the frequency of fires was too high: ≥3 fires) and played an important role in litter production. Litter cover remained homogeneous in the plot regardless of the tree cover and no correlation was found between litter cover and the cover of the other strata. In mixed stands, litter comprised a mix of *Q. pubescens* leaves and *Pinus halepensis* needles. The oak leaves formed thick well-ventilated litter in contrast to pure needle litter which was much more compacted and decomposes more slowly than sclerophyllous leaf litter (Madeira and Ribeiro, 1995; Paul and Polglase, 2004) due to a high initial lignin/nitrogen ratio (Madeira and Ribeiro, 1995).

## Conclusion

The fire regime, which is partly induced by human activities in limestone Provence, clearly influenced the structure of the stands. As diverse fire regimes within ecological limits are essential for maintaining biodiversity, the variation in vegetation structure may play a role in diversity at local and landscape scales (De Grandpré *et al.*, 2000; Burrow, 2008). In our study,

the decrease in the interval since the last fire led to the opening of the canopy similar to a level of an early successional stage. Only the cover of the lowest stratum (0-1 m), corresponding to grasses and low shrubs, did not vary as a function of the fire regime and its values were higher in shrublands whereas the cover of the other strata decreased with a decrease in the canopy cover. The horizontal connectivity between fuel beds did not change as a function of vegetation type or fire regime because of the resilience of the resprouters *Q. ilex* and *Q. coccifera* in the understorey. In this case, the risk of surface fire remains high especially when the weather conditions are severe. In contrast, the vertical connectivity decreased with fire recurrence especially in *Pinus halepensis* stands which become less resilient (Eugenio *et al*. 2006), thus decreasing the risk of crown fire. However, there was a high risk of fire in mixed stands in which no fire had occurred because of higher vertical connectivity and higher fuel amount.

The litter parameters were also affected by the recurrence of fires and by a decrease in the interval since the last fire. The increase in litter depth with stand closure increased the risk of fire. The positive correlation between litter parameters and tree cover was mainly due to the oak stratum (6-10 m) in mixed stands as oak leaves were less compacted than litter that contained only pine needles.

It will be interesting to compare these results to those obtained in acidic soils of Provence and determine whether the effects of fire recurrence on vegetation structure are the same, since the site fertility and the species composition of the fuel complex are different.

These results underline the need to control the growth of understorey vegetation to reduce the risk of fire. Prescribed burning may be an efficient technique for this purpose, but is not often applied in the French Mediterranean region.

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# TABLES

Table 1: Number of study sites sampled in each vegetation type as a function of the fire regime (number of fires/interval since the last fire).

| | Types of Vegetation | | | |
|---|---|---|---|---|
| Fire regime | Shrublands | Mixed stands | Pure pine stands | Total |
| No fire/>45 years | 3 | 4 | 4 | 11 |
| 1-2 fires/>15 years | 4 | 4 | 4 | 12 |
| 1-2 fires/<15 years | 4 | 4 | 4 | 12 |
| ≥3 fires/<15 years | 4 | 0 | 0 | 4 |
| Total | 15 | 12 | 12 | 39 |

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Table 2: Mean (standard deviation) values of the different vegetation structures as a function of the fire regime.

| Vegetation structures / Fire regimes | | No fire | 1-2 old fires | 1-2 recent fires | >=3 fires |
|---|---|---|---|---|---|
| Percentage cover of stratum | Stratum >10 m | 39.7 (37.9) | 7.1 (12.3) | 5.0 (8.7) | 0.0 |
| | Stratum 6-10 m | 11.9 (17.9) | 16.2 (14.4) | 0.2 (0.2) | 0.0 |
| | Stratum 3-6 m | 17.0 (12.9) | 17.2 (18.3) | 3.7 (5.6) | 0.0 |
| | Stratum 1-3 m | 53.6 (15.1) | 58.3 (4.7) | 45.6 (30.0) | 1.7 (2.4) |
| | Stratum 0-1 m | 70.3 (6.8) | 62.5 (19.4) | 58.7 (16.3) | 73.7 (12.5) |
| Percentage cover of structural elements | Trees | 69.7 (44.6) | 33.8 (31.7) | 10.1 (17.5) | 0.0 |
| | Shrubs | 7.0 (4.4) | 10.7 (7.5) | 17.4 (15.5) | 1.4 (1.7) |
| | Patches | 5.4 (6.6) | 16.0 (17.7) | 19.4 (12.4) | 0.4 (0.7) |
| | Layers | 52.0 (16.9) | 64.8 (6.0) | 46.8 (4.7) | 66.6 (12.5) |
| Height (m) of structural elements | Trees | 11.9 (3.8) | 8.9 (1.2) | 6.2 (1.2) | - |
| | Shrubs | 2.2 (0.8) | 2.2 (0.6) | 1.9 (0.4) | 0.6 (0.1) |
| | Patches | 1.5 (1.1) | 2.1 (0.5) | 1.4 (0.4) | 0.5 (0.1) |
| | Layers | 1.2 (0.5) | 0.9 (0.4) | 0.8 (0.4) | 0.3 (0.1) |
| Tree density (nb $ha^{-1}$) | | 394 (153) | 204 (92) | 117 (87) | 0 |
| Crown volume ($m^3$ $ha^{-1}$) | | 87121 (37934) | 24218 (20683) | 25874 (18046) | 0.00 |

Ganteaume A., Jappiot M. Lampin-Maillet C., Curt T. and Borgniet L. (2009) Fuel characterization and effects of wildfire recurrence on vegetation structure on limestone soils in southeastern France, Forest ecology and management, 258, S15-S23.
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Table 3: Mean (standard deviation) values of the different vegetation structures as a function of vegetation types

| Vegetation structures / Vegetation types | | Shrublands | Mixed stands | *P.halepensis* stands |
|---|---|---|---|---|
| Percentage cover of stratum | Stratum >10 m | 0.6 (0.8) | 13.3 (23.1) | 37.9 (34.4) |
| | Stratum 6-10 m | 0.6 (0.8) | 20.1 (17.4) | 7.7 (11.7) |
| | Stratum 3-6 m | 1.0 (1.2) | 21.6 (18.7) | 15.6 (6.3) |
| | Stratum 1-3 m | 32.3 (30.3) | 60.0 (18.0) | 53.3 (13.7) |
| | Stratum 0-1 m | 77.6 (5.9) | 58.7 (6.5) | 56.7 (15.9) |
| Percentage cover of structural elements | Trees | 4.6 (9.2) | 43.0 (45.5) | 64.4 (34.9) |
| | Shrubs | 4.3 (2.0) | 14.1 (4.5) | 15.9 (16.7) |
| | Patches | 3.5 (4.1) | 22.6 (17.1) | 13.5 (10.7) |
| | Layers | 62.8 (9.3) | 55.5 (13.7) | 46.6 (11.1) |
| Height (m) of structural elements | Trees | 9.9 (3.4) | 8.1 (2.0) | 10.5 (2.2) |
| | Shrubs | 1.6 (0.4) | 2.5 (0.8) | 1.8 (0.5) |
| | Patches | 0.7 (0.2) | 2.5 (0.9) | 1.5 (0.7) |
| | Layers | 0.7 (0.23) | 1.1 (0.5) | 1.1 (0.5) |
| Tree density (nb $ha^{-1}$) | | 15 (25) | 248 (122) | 448 (178) |
| Crown volume ($m^3.ha^{-1}$) | | 3284 (5688) | 58333 (20839) | 74502 (48240) |

Ganteaume A., Jappiot M. Lampin-Maillet C., Curt T. and Borgniet L. (2009) Fuel characterization and effects of wildfire recurrence on vegetation structure on limestone soils in southeastern France, Forest ecology and management, 258, S15-S23.
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Table 4: Results of the one-way ANOVA showing the effects of vegetation types and fire regimes on vegetation parameters (F: Fisher test ; KW: Kruskal-Wallis test ; p: probability ; NS: non significant; degree of freedom=1 in all the tests)

| Vegetation parameters | Vegetation types | Interval since last fire | Number of fires |
|---|---|---|---|
| Tree height | NS | KW=8.51; p=0.014 | KW=8.08 ; p=0.004 |
| Shrub height | F=4.20; p=0.025 | NS | KW=6.41; p=0.040 |
| Patch height | F=8.90; p=0.003 | NS | NS |
| Layer height | KW=7.48; p=0.024 | KW=13.45; p=0.004 | KW=12.96. p=0.001 |
| Tree cover | F=19.93; p=0.000 | KW=16.41; p=0.000 | KW=13.32; p=0.001 |
| Shrub cover | KW=9.33; p=0.009 | NS | NS |
| Patch cover | KW=7.32; p=0.026 | NS | NS |
| Layer cover | NS | NS | NS |
| Tree density | KW=21.13; 0.000 | KW=13.28; p=0.001 | KW=11.58; p=0.003 |
| Tree volume | KW=18.89; p=0.000 | KW=15.27; p=0.000 | KW=13.72; p=0.001 |
| 0-1 m stratum cover | F=3.40; p=0.045 | NS | NS |
| 1-3 m stratum cover | NS | NS | KW=9.12; p=0.010 |
| 3-6 m stratum cover | KW=15.30; 0.000 | KW=15.23; p=0.000 | KW=9.51; p=0.008 |
| 6-10 m stratum cover | KW=11.63; p=0.003 | KW=9.19; p=0.010 | NS |
| >10 m stratum cover | KW=10.08; p=0.006 | KW=18.12; p=0.000 | KW=18.05; p=0.000 |

Ganteaume A., Jappiot M. Lampin-Maillet C., Curt T. and Borgniet L. (2009) Fuel characterization and effects of wildfire recurrence on vegetation structure on limestone soils in southeastern France, Forest ecology and management, 258, S15-S23.
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Table 5: Comparison between the effects of the different fire regimes on strata cover and on the cover of structural elements in shrublands, mixed stands and pure pine stands (NS: non significant variation, *: significant variation, unburned: no fire, previously burned: 1-2 fires >15years ago, recently burned : 1-2 fires≤15years ago or ≥3 fires <15years ago in shrublands).

| | | Shrublands | Mixed stands | Pure pine stands |
|---|---|---|---|---|
| Strata cover | 0-1 m | NS | NS | NS |
| | 1-3 m | * Unburned or previously burned *vs* recently burned | NS | NS |
| | 3-6 m | NS | * Unburned or previously burned *vs* recently burned | NS |
| | 6-10 m | NS | * Unburned or previously burned *vs* recently burned | NS |
| | >10 m | NS | * Unburned/Burned | * Unburned *vs* Burn |
| Cover of structural elements | Trees | NS | * Unburned or previously burned *vs* recently burned | * Unburned or pr burned *vs* recently b |
| | Shrubs | NS | NS | NS |
| | Patches | NS | NS | NS |
| | Layers | NS | NS | NS |

Ganteaume A., Jappiot M. Lampin-Maillet C., Curt T. and Borgniet L. (2009) Fuel characterization and effects of wildfire recurrence on vegetation structure on limestone soils in southeastern France, Forest ecology and management, 258, S15-S23.
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# FIGURE CAPTIONS

Figure 1: Study sites and sampling in limestone-derived soils in Provence, France. Inner lines indicate municipalities (BD Carto ® IGN 2003).

Figure 2: Description of the litter in a 20m x 20m plot

Figure 3: Variation in the cover of the vegetation strata as a function of the fire regime of each vegetation type.

Figure 4: Variation in the cover of structural elements as a function of the fire regime of each vegetation type.

Figure 5: Percentage cover of the main species recorded on the study sites as a function of vegetation type and fire regime (P.h: *Pinus halepensis*, Q.c: *Quercus coccifera*, Q.i: *Q. ilex*, Q.p: *Q. pubescens*, R.o: *Rosmarinus officinalis*, J.o: *Juniperus oxycedrus*, C.m: *Cistus monspelliensis*, C.a: *C. albidus*, P.a: *Phillyrea angustifolia*, P.l: *P. latifolia*, U.p: *Ulex parviflorus*)

Figure 6: Variation in litter depth (cm) and in percentage cover as a function of vegetation type and fire regime.